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FACTORS INVOLVED
IN LAND APPLICATION
OF AGRICULTURAL
AND MUNICIPAL WASTES

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FACTORS INVOLVED IN LAND APPLICATION OF
AGRICULTURAL AND MUNICIPAL WASTES

NATIONAL PROGRAM STAFF
SOIL, WATER AND AIR SCIENCES
BELTSVILLE, MARYLAND 20705

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PREFACE

A need was expressed by the Soil Conservation Service for providing field people with information that would aid them in making recommendations for land application of agricultural and municipal wastes. Considering the great variability in waste characteristics and environmental conditions, it was apparent that a discussion of the various factors involved would be helpful in achieving a better understanding of the total problem.

Toward this end, ten specific related subjects were identified. Knowledgeable specialists for each subject were requested to write a brief non-technical discussion, including a few good basic references. The reports, which are unedited, are presented here.

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SOILS AND SITE SELECTION

By K. W. Flack and F. J. Carlisle ^{1/}

Selection of a suitable site for disposal of organic wastes on land requires consideration of a number of factors. Among these are the amount and characteristics of the waste, the physical and biological characteristics of the site, including kinds of soil and climate, the waste disposal system to be used, applicable laws and regulations, economic feasibility, and aesthetic considerations. Characteristics of the site are discussed in this chapter.

Probably no soil is perfect for waste disposal. Selection of the most suitable site generally will require weighing the advantages and disadvantages of alternative sites to select one which promises minimum damage to the environment for a particular waste disposal system in a given climatic setting. On the other hand, there can be no compromise on two important considerations in selecting a disposal site. The site must not be subject to flooding and there must be no loss of soil (adsorbed pollutants) from the disposal site.

^{1/} United States Department of Agriculture and
Soil Conservation Service

Soil Limitations for Waste Disposal

A tentative guide for evaluating soil limitations for waste disposal systems is given in Tables 1 and 2. It is assumed that a crop is to be grown and that the crop will remove most of the plant nutrients and a large part of the water applied with the wastes. The tables may also be used as general guides for evaluating soil limitations for disposal systems in which considerably more plant nutrients than can be taken up by crops are to be applied. Table 1 is for liquid wastes that can be moved by pumps and applied through sprinkler systems. This includes effluents from secondary sewage treatment plants and fluid sludges from such plants. Table 2 is for solid wastes, including sludges that cannot be moved by pumps.

Permeability, infiltration rate, soil drainage class, runoff, flooding hazard, and available water holding capacity are considered in rating soils for accepting liquid wastes and, except for infiltration rate, in rating soils for accepting solid wastes. These and related soil qualities are discussed in relation to waste disposal in succeeding paragraphs. Climate-- temperature, precipitation, length of growing season--is not included in the tables because it has little influence on site selection within small geographic areas. The importance of considering climate in the design of land-based waste disposal systems is discussed in later sections.

Soil Moisture Relationships

Desirable moisture relationships for land-based disposal systems are complex and are not amenable to generalizations. The requirements are influenced by the kind of waste, whether liquid or solid, and by the

objectives of the system. There are situations that are clearly undesirable; for example, soils that are both poorly drained, and very slowly permeable have severe limitations for year-round disposal systems. On the other hand, one cannot establish ideal site requirements for all situations. The amount, kind, and distribution of precipitation relative to evapotranspiration and the requirements of the disposal system must be considered. Solid organic waste needs enough moisture to enable microorganisms to function and, if the site is to be vegetated, for plants to grow. There must also be enough moisture in addition to that lost through evapotranspiration, to leach salts beyond the root zone. Yet, rapid movement of large volumes of water through the soil is undesirable. For liquid waste disposal systems, the requirements are different. A large part or all of the water needs of the microorganisms and vegetation will be supplied by the waste water. A large excess of water beyond that used for evapotranspiration may be undesirable, particularly if economic returns from the use of waste water for irrigation are expected to offset part of the cost of the system or, at least, are expected to pay the farmer for the trouble and inconvenience of using waste water. Obviously, potential benefits from waste water are greater in areas where evapotranspiration greatly exceeds precipitation. Some precipitation however, will be useful to help leach salts that may otherwise accumulate in the root zone. These are important considerations in deciding whether disposal on land is feasible and in defining site and soil requirements for a particular situation.

A large water holding capacity is important for solid waste disposal and for liquid waste disposal if the soil must be able to accept precipitation after it has been irrigated with waste waters. A high water holding capacity is least important in arid areas provided that the waste can be applied frequently and rates as well as timing can be adjusted to evapotranspiration while maintaining desirable leaching rates. Minimum available water-hold capacity requirements in Table I have been introduced primarily as a measure of a desirable minimum soil volume that is necessary to complete reactions within the root zone.

Very rapid or slow permeability (Table I) is undesirable. If the permeability is slow, loading rates for liquid wastes are necessarily low. If the soil is very rapidly permeable, the water moves through the root zone so quickly that little of the water and the plant nutrients it contains can be taken up by plants. Moreover, soluble organic matter may not be decomposed and nitrates not be reduced to gaseous form. Most, although not all, very rapidly permeable soils also have limited absorptive capacity.

Infiltration rate of water into the soil controls the rate at which liquid wastes can be applied without causing runoff. As discussed elsewhere, infiltration rate is much affected by soil management practices.

The conditions described in the preceding paragraph may be induced in some soils through water management. Soils having moderately or moderately rapid permeable horizons in the upper part of the section overlying more slowly permeable subsoil horizons should be most amenable to management for those conditions. Nitrogen transformations and the hydraulic aspects are discussed in subsequent chapters.

Poorly drained soil may be difficult to manage if the waste is to be transported by trucks or if the land is to be farmed, because the soil may not be accessible to vehicles during much of the year.

A potential for rapid runoff is undesirable for two reasons: When organic waste is placed on the surface it may be carried directly into surface water. Also, potentially rapid runoff markedly increases the hazard of soil erosion. Erosion can carry soil particles with sorbed wastes away from the disposal site and into surface water. Slope, infiltration rate, permeability of subsurface layers, and temperature (frozen soil) are soil properties or qualities that affect potential runoff.

Adsorptive and Filtering Properties of Soils

The capacity of a soil to absorb components of wastes depends on the physical and chemical characteristics of the soil material and the volume of soil available. The many kinds of chemical reactions between soil and waste materials and the ability of soil to act as a physical filter are discussed elsewhere. They depend on mineralogy, porosity, surface area, presence of amorphous or semicrystalline sesquioxides, the presence of free carbonates, the redox potential,

and similar soil properties. Most of the chemical reactions can only take place within a relatively narrow pH range and a considerable buffer capacity of the soil is therefor desirable. Amounts of highly divided iron and aluminum oxides which are important in phosphate fixation processes in acid soils are generally correlated with clay content; fine textured soils contain more of such compounds than coarse textured soils. The calcium-based phosphate fixing mechanism in neutral and alkaline soils is less related to texture if calcium carbonate is present.

The effective depth of soils varies widely. Many soils are underlain at shallow depths by materials that either do not transmit water or that have little or no capacity to filter or absorb. In some soils, however, such as those in thick deposits of alluvium, glacial drift, or lacustrine material, or very deeply weathered old soils in warm humid areas, the effective depth as far as the absorption or filtering of waste is concerned may extend far below the soil proper and may be tens or even hundreds of feet. At the other extreme, notably in acid, sandy soils, the chemically active material may only be a few inches thick. Unless precautions are taken, this layer may be easily destroyed in site preparation. In any case, these soils may have very limited loading capacity. Inasmuch as the sandy soils are desirable from an engineering viewpoint because of their high potential infiltration rates and rapid permeability, soils of this kind have frequently been selected for disposal systems.

Available water capacity is in a general way related to the total absorptive capacity of the soil. Most soil materials that have a high available water capacity also have a relatively high capacity to absorb

waste; conversely, soil materials having a very low available water capacity also have a low absorptive capacity. Available water capacity, thus, is used in Tables I and 2, not only because of its importance to the hydrology of the system, but also because of its relationship to the absorptive capacity of the total soil volume. Moreover, estimates of available water holding capacities of soils generally are included in modern soil surveys and so are commonly available.

Climatic Factors in Site Selection

It has been mentioned that when selecting a site for waste disposal in a local area, one has little choice with respect to climatic factors. At the same time, temperature, amount and distribution of precipitation, and length of the growing season strongly affect the design and the feasibility of land-based waste disposal systems. Suitable temperature and moisture conditions are necessary for high rates of decomposition of wastes by microorganisms. The relationships are discussed in another section of this publication. Also, relatively little water is removed from the soil by evapotranspiration during the cold season. Limitations imposed by the cold season can be overcome but the cost increases with increasing intensity and length of the cold season.

Two ideas are emphasized in this section. First, a soil that receives much water from rainfall commonly cannot handle a lot more in the form of liquid waste. The liquid waste may overtax the absorptive capacity of the system if the extra water is added to a large natural precipitation. Secondly, the design of land-based disposal systems cannot depend on

average climatic conditions. One unfavorable year in 8 or 10 may critically limit the system.

In designing a system, the total amount of precipitation relative to evapotranspiration, the distribution of precipitation during the year, and the changes in precipitation from year to year have to be considered. Figure 1 shows the excess of mean annual potential evapotranspiration as estimated from Thornthwaite's formula over mean annual precipitation for the United States. The values are in inches. Positive values, in the western part of the country, indicate excessive evapotranspiration as high as 70 inches. Obviously, the higher the excess of evapotranspiration the more waste water can be added to the land without causing leaching and the greater is the value of the waste water for irrigation. To the east of a line running slightly to the west of the Mississippi River, negative values suggest some runoff and leaching even if no waste water is added to the soil. Actual excesses and deficits depend, largely, on the distribution of precipitation throughout the year. Even in the humid part of the country, there is usually a moisture deficit during part of the summer. There may be an excess of moisture during the winter in some of the more arid parts of the country.

Variations from year to year must also be considered. A system that works well in an average year may be seriously overloaded during an exceptionally wet year. Figure 2 shows the frequency distribution of precipitation deficits for six weather stations in the midwest. The range between the effectively driest 5 percent and the wettest 5 percent of the years for individual stations is between 30 and 55 inches. The range of 55 inches is the same order of magnitude as the differences between the extremes in mean annual values over roughly four-fifths of the country (Figure 1).

Thus, the combined effects of variations in precipitation and potential evapotranspiration from year to year that need to be considered in designing a system for a particular location may be about as large as the difference over much of the country when the latter are in terms of mean annual values. The disadvantages of erratic precipitation can partly be compensated by proper system design and by restricting disposal sites to the most favorable soils.

Other Considerations

Areas having consolidated rock at shallow depths are undesirable for waste disposal because the reduced volume of soil material limits the absorptive and filtering capacity of the site. Areas having jointed igneous rocks or limestone with solution channels at shallow depths are especially poor sites for waste disposal because of the hazard that polluted water may move through the cracks and channels in the rock and into the ground water. The nature of the rock or regolith below depths of five or six feet is more important for disposal systems with high-loading rates than for systems with low-loading rates.

Waste disposal systems commonly are applied on land areas large enough that soil conditions are not the same everywhere within the area. Adjacent soils may differ significantly in one or more of the soil qualities listed in Tables 1 and 2. The design of the disposal system and the soil and plant management systems used obviously must take such variability into consideration.

Soil Surveys

Of the sources of information that are commonly available for making initial judgments regarding the suitability of potential land-based waste

disposal sites, soil surveys probably are the most useful. The data of soil surveys may be used for this purpose in two somewhat different ways. In the absence of experience on the performance of particular kinds of soil for waste disposal, one relies on the general body of knowledge of soil science to estimate how particular kinds of soil will behave if used for a given waste disposal system. Judgments are based on knowledge of the relationships between individual soil properties or sets of a few properties and site characteristics that are desirable for waste disposal systems. A common denominator for adding up the effects of many relevant properties in a given set is lacking. Predictions of soil behavior commonly must be in general terms. The state of knowledge for making such evaluations is here and in other sections of this publication.

Soil surveys are especially useful as a basis for transferring experience on a named kind of soil at one place to the same kind of soil at another location. Quantitative predictions of soil behavior are possible on this basis. The accuracy of such predictions depends on the relevance of the soil properties that have been used in the classification of the soils and on the validity of the correlation.

Modern detailed soil surveys provide information on many soil properties and qualities for particular places. Among those that are of consequence for waste disposal systems are: soil drainage class; estimates of soil permeability and of available water holding capacity; soil slope and other soil characteristics that affect runoff; soil reaction (pH); estimates of flooding hazard; soil texture; mineralogical composition in broad groups; soil temperature and soil moisture regimes in broad classes; the presence or absence of soil horizons or layers that restrict permeability to plant

roots, water, and air; estimates of engineering properties of soils; and for drier parts of the country, salinity. Such information is provided for kinds of soils--usually soil series and phases of soil series--for which mapping units delineated in the soil survey are named.

The most specific information related to the suitability and limitations of soils for waste disposal systems is in soil surveys at scales of 1:20,000 or larger, in which most mapping units are named as phases of soil series. Such surveys are useful for preliminary or initial evaluation of specific tracts of land. They can be used to greatly narrow the number of alternative sites and lead to those that show most promise for the objectives. Such soil surveys indicate the kinds of limitations at prospective sites and provide information pertinent to the design of disposal systems and the application rates most likely to result in satisfactory performance. Final decisions on suitability of the land and design of the disposal system must be preceded by onsite inspection and investigation of individual tracts.

General soil maps at smaller scales, 1:63,360 or smaller, usually are helpful to identify areas where some proportion of the area has a particular potential or degree of limitation for waste disposal on the land. Such maps, however, generally are not suitable for evaluating small tracts of land.

Identification of kinds of soil in classes in the higher categories of the taxonomic soil classification system commonly provides information that is relevant to selecting sites and designing systems for waste disposal on land. The names of classes in the higher categories of the soil

classification system are descriptive if the reader understands the meanings of the formative elements of the names. Many inferences can be drawn from classes in various levels of the system but only a few examples can be set down here. The few examples will indicate some of the kinds of information that can be inferred from the class names.

Among the suborders of Mollisols, Borolls have long, cold winters when the soil will be frozen for extended periods and the time that the soil is warm enough for high rates of biological activity is short; Zerolls are dry for extended periods in summer, but in most of them, moisture moves through the soil and into underlying layers in winter; Aquolls are naturally wet and often develop in low places where water collects and stands.

Among the great groups of Xerolls, Calcixerolls have a strong concentration of calcium carbonate or gypsum at some depth above 60 inches, and the soil is calcareous in all parts above that depth unless the texture is sandy; Argixerolls have an horizon of clay accumulation in the subsoil and most have an horizon of carbonate accumulation below that.

Among the subgroups of Argixerolls, Aquic Argixerolls have moderately shallow ground water at some time of the year unless artificially drained, Lithic Argixerolls have hard rock at shallow depths, and Vertic Argixerolls are fine-textured soils that have deep, wide cracks at some period in most years. Typic Argixerolls lack the features or combinations of features specified for other subgroups.

Explanations of the meanings in the class names and definitions of classes are given in "Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys" (in press).

The concept of soil drainage class involves the permeability of the soil and ground water relationships that are controlled by factors outside the soil. Soil drainage class as used here refers to the depth to a water table and the proportion of time that the water table is present when the soil is warm enough for biological activity. The water table may be controlled by the properties of the soil, its position on the landscape, or other factors that control the regional ground water level.

Relatively well drained soils are desirable for disposal of organic waste because the organic matter decomposes rapidly and nitrates are formed under the prevailing aerobic conditions. Nitrates formed in amounts larger than can be used by microorganisms and plants, may be leached through the soil and into the ground water. In poor and very poorly drained soils in which the ground water is near the surface of the soil during large parts of the year, organic wastes in the soil may not be oxidized, and undesirable soluble organic and inorganic compounds may form and move into the ground water.

In soils of intermediate drainage, organic matter may be decomposed and inorganic compounds precipitated in the aerobic upper part of the soil while nitrates may be reduced to nitrogen gas in the saturated subsoil where reducing conditions prevail. This mechanism for reducing nitrate works, however, only if the soil is warm enough to be biologically active and if reducing conditions exist at relatively shallow depths where organic matter is available as a source of energy for the reducing bacteria.

POTENTIAL EVAPOTRANSPIRATION MINUS MEAN ANNUAL PRECIPITATION (inches)

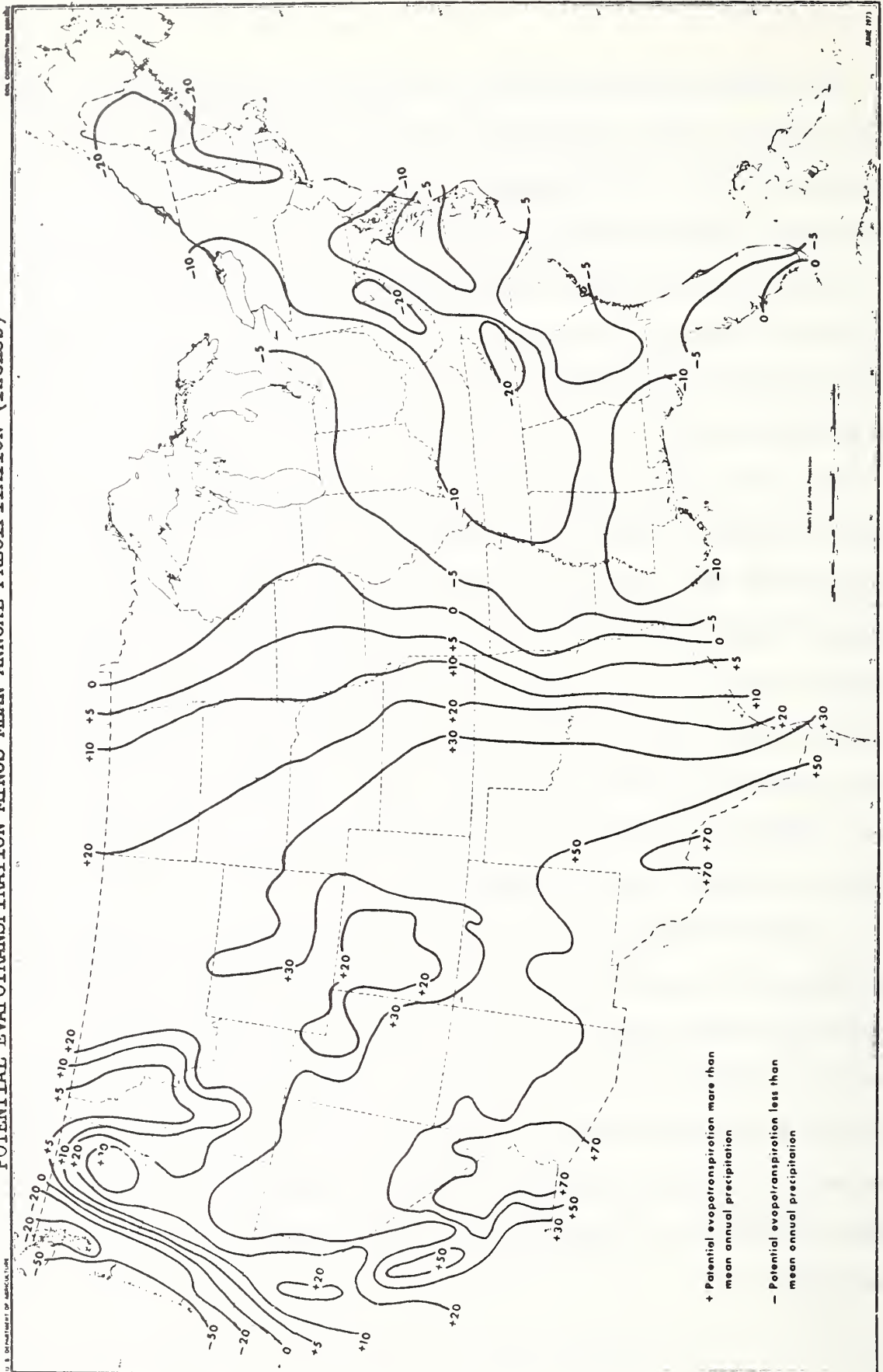
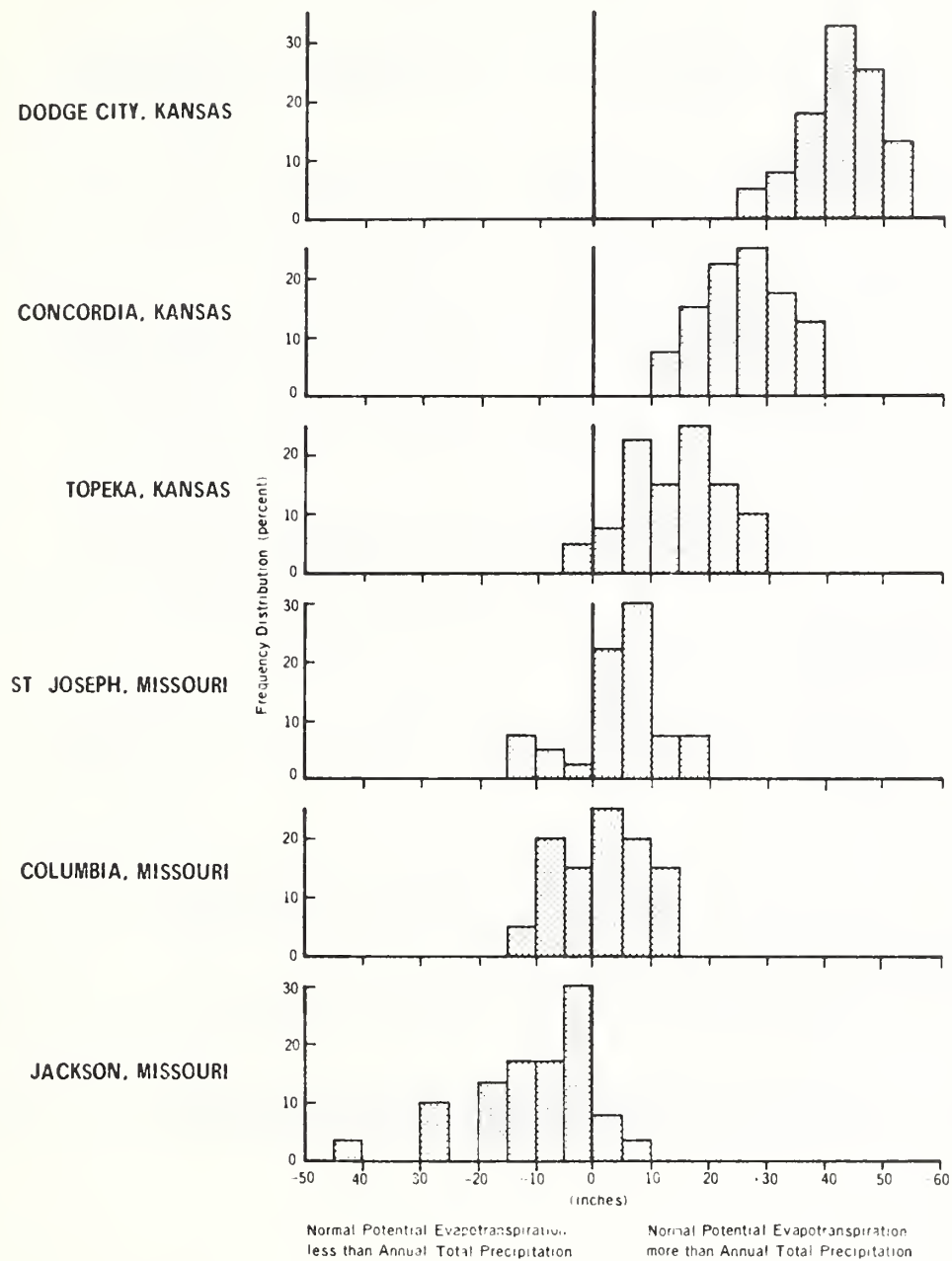


Figure 1

FREQUENCY DISTRIBUTION

Normal Potential Evapotranspiration Minus Annual Total Precipitation



JUNE 1973

Figure 2

Table 1.--Soil Limitations for Accepting Nontoxic Biodegradable Liquid Waste^{1/}

Item Affecting Use ^{2/}	Degree of Soil Limitation		
	Slight	Moderate	Severe
Permeability of the most restricting layer between 60 inches and surface horizon	Moderately rapid and moderate 0.6-6.0 in/hr	Rapid & modert. slow-6-20 and 0.2-0.6 in/hr	Very rapid, slow, and very slow >20 and <0.2 in/hr
Infiltration Rate	Very rapid, rapid, moderately rapid, and moderate ≥ 0.6 in/hr	Moderately slow 0.2-0.6 in/hr	Slow and very slow < 0.2 in/hr
Soil Drainage Class	Well drained and moderately well drained	Somewhat excess. drained and somewhat poorly drained	Excessively drained, poorly drained, and very poorly drained
Runoff	None, very slow, and slow	Medium	Rapid and very rapid
Flooding	Soil not flooded during any part of the year	Soil flooded only during nongrowing season	Soil flooded during growing season
Available Water Capacity from 0 to 60 inches or a limiting layer ^{3/}	Temporary Installation → 7.8 Permanent Installation → 3 inches	3-7.8	< 3 inches < 3 inches

^{1/} Modified from a draft guide dated April 27, 1973, for use in the Soil Conservation Service, USDA. Liquid wastes are those that can be moved by pumps and applied through sprinkler systems.

^{2/} For definitions see Soil Survey Manual, U.S. Department of Agriculture Handbook 18-1951.

^{3/} Available water capacity, as used here, is the difference between the amount of soil water at field capacity and the amount at wilting point.

Table 2.--Soil Limitations for Accepting Nontoxic Biodegradable
Solids and Sludges 1/

Item Affecting Use <u>2/</u>	Degree of Soil Limitations		
	Slight	Moderate	Severe
Permeability of the most restricting layer above 60 in.	Moderately rapid and moderate 0.6-6.0 in/hr	Rapid & modert. slow <u>2/</u> 6-20 and 0.2-0.6 in/hr	Very rapid, slow, and very slow > 20 and < 0.2 in/hr
Soil Drainage Class	Well drained and moderately well drained	Somewhat excess. drained & somwh. poorly drained	Excessively drained, poorly drained, and very poorly dr.
Runoff	None, very slow, and slow	Medium	Rapid and very rapid
Flooding	Soil not flooded during any part of the year		Soil flooded during some part of the year
Available water capacity from 0 to 60 inches or to a limiting layer <u>3/</u>	> 7.8 inches	3-7.8 inches	< 3 inches

1/ Modified from a draft guide dated April 27, 1973, for use in the Soil Conservation Service, USDA. Solid wastes are those that cannot be moved by pumps.

2/ For definitions see Soil Survey Manual, U.S. Dept. of Agriculture Handbook 18- 1951.

3/ Available water capacity, as used here, is the difference between the amount of soil water at field capacity and the amount at wilting point.

Waste Characteristics

J. D. Menzies and R. L. Chaney ^{1/}Introduction

In considering the characteristics of wastes to determine their suitability for application to land three premises are adopted: (1) the application is beneficial to the normal use of land; (2) public health is not adversely affected; and (3) the air and drainage water are not polluted. Normal use of land includes agriculture, parks, forests, and landscaped developed areas. Obviously land can be used as disposal or storage space as in lagoons, stockpiles or dumps but this use of land is outside the scope of this report. Generally, wastes that are potentially compatible with normal land use are decomposable organic wastes containing N, P, K and low levels of other elements. For this reason industrial wastes, such as metallurgical factory sludges, stack scrubbing sludges, and ash residues, are not included here. Thus we are talking about animal, municipal, and agricultural processing wastes and their suitability for incorporation into agricultural land in a beneficial way.

Animal Manures

Animal feeding operations are often concentrated into large units often close to population centers. This has changed the problem from one of insufficiencies of on-farm manure to one of surplus for the readily available land. This raises the question of how much can be

1/ Laboratory Chief and Plant Physiologist, respectively, Biological Waste Management Laboratory, Agricultural Environmental Quality Institute, Northeastern Region, Agricultural Research Service, U.S. Department of Agriculture.

safely added to land rather than how much is needed to maintain productivity. Under these circumstances it is necessary to know much more about the composition of the manure products being used.

In Tables 1, 2, and 3 some average values or normal ranges of manure constituents have been compiled from a variety of sources. Manure characteristics differ widely among classes of animals and even from the same kind of livestock under different feeding regimes or different manure handling systems. In making recommendations on high loading rates it is important to know whether the manure in question is typical or whether it contains unusual amounts of some additive such as pesticide residues or salt.

Manures in general are high in organic matter and low in inorganic residues. They contain sufficient nitrogen to permit rapid microbial breakdown of the readily decomposable organic matter, leaving adequate mineralized nitrogen for crop use and potentially excess nitrogen which may leach to groundwater. The organic matter of manure is less persistent than that found in well digested sewage sludge. Historically it has been found necessary to keep adding manure to land to maintain any significant increase in soil organic matter beyond the normal level for the soil and climate. Thus the organic loading on soil from manure is not a limiting factor. Microbial processes will quickly reduce this to a small, resistant residue with an accompanying improvement in soil structure.

The nitrogen content of manures is usually the most important limiting factor in land loading rates. As seen in Table 1 farm animal manures will have nitrogen contents in the range of 2 to 4 percent on a dry weight

basis. Fresh manure can be expected to have a moisture content of 70 to 80% unless diluted with water from washing operations or rainfall. Therefore a crude rule is to assume that manure will run 10 to 20 lbs. N per ton of fresh manure. As the loading rate per acre is increased above a normal 10 to 20 wet tons per acre the possibility of nitrate leaching to groundwater increases. At very high rates (80-100 tons wet weight/acre) there may be nitrogen loss due to ammonia volatilization, or denitrification if the soil is wet, but this may not be sufficient to counteract excess nitrification. Another hazard of high nitrate level in soil is that forage crops may contain nitrate levels hazardous to livestock. Best farming practice would dictate that manure applications be limited to rates supplying no more mineral nitrogen than the crop can use. Under most circumstances this rate will be less than 30 to 50 tons per acre fresh weight.

Beef cattle feed-lot-manure contains excess sodium because of the practice of providing large amounts of sodium chloride in the feed. Salt accumulations may damage crops if high rates of this manure are used on areas where little or no leaching occurs. Salt problems may be particularly serious when feed lot lagoon water is spread on land for disposal. Under these circumstances salt problems may develop before the nitrogen becomes excessive. The salt not only is toxic to plant growth, especially seedlings, but also disperses soil aggregates and reduces water infiltration rates. Studies in Nebraska have shown that feed lot manure may contain from 500 to 1600 ppm Na on a dry weight basis (equal to 2.5 to 8 lbs. NaCl per ton).

Minor and trace elements such as S, Ca, Mg, Fe, B, Cu, Co, Zn, Mn and Mo are present in farm animal manures (Tables 2 and 3). These normally are never present in sufficient quantities, even at very heavy manure rates, to be anything but beneficial for crop production. Occasionally Cu and Zn may be added to swine or poultry feed at up to 250 and 100 ppm, respectively. The animals digest 80-90% of the diet, and excrete nearly all of the added Cu and Zn. Thus, the manures could contain 1250-2500 ppm Cu and 500-1000 ppm Zn. These levels of Zn and Cu in manure comprise a potential toxicity problem similar to that from sewage sludge. Because the application rate for swine manure is limited to 5-10 dry tons/Acre, because of its N content, metal buildup in the soil will occur over many years.

Pesticide residues in manures can occur if these compounds are used in the feed or around the animals. These concentrations, however, can be expected to be so low that there is no danger of plant damage or plant uptake to levels exceeding established tolerances. There is always some risk, however, under unusual circumstances, or with new pesticide materials, that pesticide residue damage may occur. The use of the most potent persistent pesticides is being restricted so they should cease to be any problem in manure use on land. Conceivably continued use of arsenicals or copper in feeds could lead to plant toxicity problems under repeated heavy use of manure but to our knowledge this has not so far occurred. The best practice is to keep alert for any reports of pesticides or feed additives affecting the use of manure on land.

Farm manures always present some hazard in dissemination of livestock disease organisms. Certain well known quarantinable pathogens such as that of hog cholera or avian Newcastle disease require destruction of contaminated manure (See Chapter on Pathogen Considerations). General farm experience, however, indicates that the hazard to livestock health from use of manure on cultivated land or even pasture is not great, and certainly less than the hazard of animal-to-animal spread in confined areas.

Agricultural Processing Wastes

Liquid or solid wastes from agricultural processing plants or from harvesting operations are frequently returned to land both for disposal and for land improvement. Crop wastes consist of unusable plant materials generated at all stages of processing from harvesting to final utilization. Characteristically one type of waste will accumulate in a region or at a processing plant so that large tonnages of waste or large volumes liquid effluent have to be dealt with. Examples are pea vines, sunflower hulls, sugarcane bagasse, grain straw, fruit pomace, potato starch, and paper plant residue.

Most plant crop wastes are 90 percent organic, and generally low in nitrogen and other nutrient elements. They present no hazard to soil or crops so far as toxicities are concerned. Large tonnages can be disposed of by land incorporation without harm. If they are almost entirely carbohydrate or lignified material, without significant protein content, they will require added nitrogen to assure their decomposition without depriving the crop of needed nitrogen. This nitrogen requirement will be highest

for the more readily decomposable wastes and may be in the range of 1 percent of the dry weight added. This will be less if the waste decomposes slowly because the nitrogen will be recycled as the waste breaks down. It will also be dependent on the nitrogen content of the waste. (See Chapter on Nitrogen Cycle Considerations.) In some low fertility soils extra P, K, and sulfur may be needed to handle decomposition of heavy rates of such wastes.

Sewage Sludge

As other ultimate disposal options are being closed to protect our streams, the ocean, and the air, land application of sewage sludge is being considered as a viable alternative. The EPA and others have estimated that if all sewage were treated to the best of present technology, about 5 million dry tons of digested sludge would be produced in the U.S.A. each year. The sludge can be distributed as a liquid of 2-10% solids, as a "filter cake" semisolid of 18-25% solids, as sand bed dried solid of 50-80% solids, as heat-dried sludge of 94-99% solids, or sludge compost of 40-70% solids.

Sewage sludges vary widely in composition. These differences are produced by inherent variation in the wastes arriving at the sewage treatment plants, and the methods used therein to treat the wastewater. Some cities have brewery or other industries which discard wastes that are high in nitrogen (thus Milorganite can be produced at 6% N). Some cities have industries which generate large quantities of wastewater high in heavy metals. Digested sludge contains about double the P and

metal content of raw sludge because the unstable organic component is destroyed during digestion. Chemical treatment with Al^{3+} , Fe^{3+} , or lime to remove phosphate from wastewater will substantially enrich the P content of sludges compared to the 1-2% usually seen now. Use of different sewage treatment practices on the same influent sewage would lead to considerably different nutrient and heavy metal contents in the sludges produced. Thus, methods of treatment and types of wastes can produce extreme variation in the composition of sewage sludge. Each will need to be characterized by analysis.

Macronutrients: The N, P, and K contents of sewage sludge have been described by many authors. Table 4 is a general summary of the range observed. Activated sludges are high in organic N content compared to digested sludges or contain little soluble $\text{NH}_4\text{-N}$. Liquid digested sludges are higher in soluble $\text{NH}_4\text{-N}$ (often 50% of total N) compared to filter cake digested sludge (often 15-20% of total N) because much of the soluble $\text{NH}_4\text{-N}$ stays with the liquid phase during vacuum filtration or is removed during "elutriation." The soluble $\text{NH}_4\text{-N}$ can be volatilized from liquid sludge by aeration after lime treatment to high pH. The availability of the N in sludges will vary widely as related both to the amount and chemical forms of N present, and type of sludge. Digestion of raw primary or activated sludges stabilizes the sludge N considerably. Composting would be expected to stabilize this sludge N even further.

The availability of P in sewage sludges generated by different sewage treatment processes is relatively uninvestigated. The newer chemical P removal processes generate Al PO_4 and Fe PO_4 of unknown plant availability. Heat drying may markedly reduce the plant availability of P in these sludges.

The K content of sludges is the out-of-balance macronutrient. The soluble K usually remains with the water during treatment. Liquid sludges thus contain somewhat greater K content than filter cake, but the K is still low; use of sludge as a fertilizer will likely require supplementing with K fertilizers based on soil test results.

Organic Matter: The organic matter content of sludges varies from 25-80% dry weight. Again this is due to sewage source and treatment processes. Cities with combined sewers, rather than with separate sanitary plus storm sewers, generate sludges with greater sand and silt content. Activated sludge has a high organic matter content. However, the sludge organic matter varies from the highly stable humic materials of well digested sludges to the highly unstable raw activated sludges. Fat and grease contents vary and are decreased by digestion.

pH: The pH of sludges usually varies from 5.5 to 12.5. Heat-dried activated sludge ranges from 5.5 to 6.5. Unlimed digested sludge is usually pH 6.5-7.5. Sludge is often limed to pH 10-12.5 using $\text{Ca}(\text{OH})_2$ to improve vacuum filtration efficiency or to kill pathogens. When these high pH sludges are mixed into soil, however, the pH rapidly falls to 8. A new sludge chlorination treatment produces sludges of pH 2-3.

The lime equivalent of sewage sludge varies with the amount of lime added during sewage treatment or sludge filtration. Because sewage sludge amended soil should be kept about pH 6.5 to reduce availability of heavy metals, sludge with a reasonable lime equivalent is desirable.

Trace Elements: The metals Zn, Cu, Ni, and Cd are the principal minor elements in sewage sludge of interest to agriculture because of their potential toxicity to crops or consumers of crops. Berrow and

Webber (1972) have described British sludges. Many U.S. sludges have been analyzed by numerous investigators and the ranges found for digested sludges are shown in Table 5. Undigested sludges would have about 50% lower Zn, Cu, and Cd as described earlier. Several industries have been extreme metal polluters: viscose rayon (Zn), vulcanized rubber or paper (Zn), metal plating (Cu, Ni, Cd, and Cr), etc. Page (1974) has recently summarized the published data on metal content of sludges.

The analyses of these metals in specific sludges are presented in Table 5 to show the variation in toxic metal pollution patterns due to specific industrial pollution. Sludge A is contaminated by Zn from viscose rayon processing. Sludges B, C, and D are contaminated mostly from electroplating. Sludge E is contaminated by Cu wire manufacturing. Sludges F and G are relatively uncontaminated and sludge G may represent the lowest values attainable in digested sludges because of the nature of urban construction and living patterns.

The possible hazards of these metals to crops or consumers, and management practices required to control these hazards are described in detail in another chapter. In consideration of these hazards, toxic metal maximums for recommended use in agriculture are listed under "domestic" sludge in Table 6. These are based on the attainable levels which can be reached by enforcement of current regulations governing industrial waste inputs to metropolitan sewer systems. They are further based on a Benefit:Risk judgement where N, P, K, Ca, Mg (to a certain extent B, Cu, Zn) and organic matter are Benefits, and the toxic metals are Risks. There is no reasonable justification for agriculture to accept sludges with metal contents higher

than the "domestic" sludge level because this would unduly limit the total amount of sludge used on a unit of land for fertilizer benefit. Parkland, forests, land unlikely to grow food-chain crops, and land dedicated to receiving toxic sludges may not have to use these constraints.

Analyses of sludges should be furnished by municipal authorities to all parties involved in agricultural use of sewage sludges. Analyses on semi-monthly composite samples should include: N(Total), $\text{NH}_4\text{-N}$, P, Ca, Mg, Zn, Cu, Ni, Cd, Pb, Hg, Cr, B(Total or hot water soluble), soluble salts, % solids, % volatile solids (or % organic C), pH, and CaCO_3 equivalent. All methods used in the analyses should be stated.

The composition of many sludges can be expected to change in the near future. P should increase as more chemical P removal technology is used. N and metals will increase because more efficient P and BOD removal also will trap more N and metals. Metals should be decreased as the EPA standards for metals in effluent are enforced and the cities track down their metal polluters. Cities will also decrease metal pollution if agriculture demands this for safe agricultural use of the sludges. Cadmium pollution will have to be diminished in many cities before their sludges are safe for beneficial use on agricultural land. Because the cities may be able to save considerable expense by changing to land disposal, they can afford to enforce metal pollution standards, and thereby furnish agriculture with safer, hence, more beneficial sludges.

Composition of Sewage Effluents

Land disposal of secondary sewage effluent is being considered as an alternative to advanced physical-chemical technology to prepare the wastewater for release to surface waters. Table 7 shows the range of contents of macronutrients and trace elements. Ca, Mg, and K are controlled by the municipal water source and vary widely. Usually effluent is applied at sufficient rates to supply crop N requirements; P supplies then exceed crop utilization and P accumulates. The trace elements will build up slowly over many years, even centuries, before they comprise a potential phytotoxicity hazard which requires management.

Table 1. N, P, K, Content of Animal Manures^{1/}
Dry Weight Basis

Source	N		P		K	
	%	Range	%	Range	%	Range
Dairy cattle	3.3	(1.9-5.5)	.35	(0.1-0.4)	2.0	(1.0-3.0)
Beef cattle	2.0	(1.5-4.0)	.65	(0.3-0.7)	1.6	(1.0-3.0)
Hogs	2.8	(4.0-7.5)	1.0	(0.2-1.5)	1.2	(0.2-1.6)
Horses	2.5		.25		.8	
Sheep	3.5		.55		1.7	
Broilers	3.5	(1.8-6.8)	2.1	(0.5-3.2)	1.7	(1-2.9)
Laying hens	3.6		1.3		1.3	
Ducks	2.6		.8		.5	
Geese	3.3		.4		.6	
Turkeys	5.0		.6		.8	

^{1/} Average of values reported by numerous authors. Manure varies widely in moisture content, with the season, and with the composition of the diet.

Table 2. Minor Elements in Animal Manures (Loehr, 1968) Percent -
Dry Weight Basis

Source	S	Ca	Fe	Mg
Dairy cattle	.2	1.3	.02	.5
Beef cattle	.4	.6	.02	.5
Hogs	.5	2.3	.11	.3
Horses	.2	2.0	.03	.4
Sheep	.3	1.7	.05	.5

Table 3. Trace Element Concentrations in Farm Manures (Atkinson, et al. 1954) ^{1/}

Element	ppm - dry wt. basis		Average
	Minimum	Maximum	
Boron	4.5	52.0	20.2
Manganese	75.0	549.0	201.1
Cobalt	0.25	4.70	1.04
Copper	7.6	40.8	15.6
Zinc	43.0	247.0	96.2
Molybdenum	.84	4.18	2.06

^{1/} Data from manure samples from cow, horse, swine, sheep and mixed manures. See text for data on Cu and Zn in some swine and poultry manures.

Table 4. Range of N, P, and K Content of Sewage Sludge (dry weight basis)

Analysis	Observed Range	Typical
N Total	1-6%	2
NH ₄	12-55% of total N	
P as P	0.8-6%	1
as P ₂ O ₅	2-14%	
K as K	0.1-0.5%	0.2
as K ₂ O	0.1-0.6%	

Table 5. Composition of Digested Sewage Sludge from Several Cities Showing Excessive Levels of Specific Heavy Metals (dry weight basis)

City	Zn ppm	Cu ppm	Ni ppm	Cd ppm	Cd % of Zn
A	15700	230	170	61	0.4
B	5100	1540	345	320	6.3
C	20000	2110	7850	157	0.8
D	7400	390	1450	2850	38.5
E	1025	16800	145	10	1.0
F	1610	700	68	13	0.8
G	540	240	33	4	0.7

1/ Some of these analyses were provided by P. M. Giordano, T. D. Hinesly, B. D. Knezek, and T. M. Regan.

Table 6. Range of Metal Contents in Digested Sewage Sludges
(dry weight basis)

Analysis	Observed Range	"Domestic" Sludge ^{1/}
Zn	500-50,000 ppm	<2000 ppm
Cu	250-17,000 ppm	<1000 ppm
Ni	25-8,000 ppm	<200 ppm
Cd	5-2,000 ppm	<15 ppm
Cd	0.1-40% of Zn	<1.0% of Zn
B	15-1,000 ppm	<100 ppm
Pb	100-10,000 ppm	<1000 ppm
Hg	<1-100 ppm	<10 ppm
Cr	50-30,000 ppm	<1000 ppm

^{1/} Typical sludge from communities without excessive industrial waste inputs, or with adequate abatement.

Table 7. Composition of Sewage Effluent

Element	Range mg/l	Typical	
		Concentration mg/l	Application ^{1/} kg/ha/year
N	15-40	25	500
P	0.5-40	10.	200
Cd	<.005-6.4	<.005	<0.1
Co	<.05-.05	<.05	<1.0
Cr	<.05-6.8	.025	0.5
Cu	<.02-5.9	0.10	2.0
Hg	<.0001-0.125	0.001	0.02
Ni	<.02-5.4	0.02	0.4
Pb	<0.2-6.0	0.05	1.0
Zn	<.02-20.	0.15	3.0

^{1/} Applied at 5 cm/week for 40 weeks/year.

REFERENCES

1. Atkinson, A. J., G. R. Giles, and J. G. Desjardins. 1954. Trace element content of farmyard manure. Can. J. Agr. Sci. 34:76-80.
2. Berrow, M. L. and J. Webber. 1972. Trace elements in sewage sludges. J. Sci. Food Agr. 23:93-100.
3. Loehr, R. C. 1968. Pollution implications of animal wastes - a forward-oriented view. USDI, FWPCA. Reprinted 1973 as EPA Water Poll. Cont. Res. Ser. 13040, 175 pp.
4. Page, A. L. 1974. Fate and effects of trace elements in sewage sludge when applied to agricultural lands. A literature review study. Report No. EPA-670/2-74-005, 108 pp.
5. Peterson, J. R., T. M. McCalla, and G. E. Smith. 1972. Human and animal wastes as fertilizers. Fertilizer Technol. and Use, 2nd Ed. 557-596.

PATHOGEN CONSIDERATIONS

W. D. Burge 1/

This chapter is concerned with disease hazards associated with land application of animal, sewage, and food processing wastes. There is little, if any, danger of dissemination of diseases from the application of food processing wastes to agricultural lands. The application of animal and sewage wastes, however, is of concern, because many of the agents of disease that afflict man and other animals, occur in excreta. In fact, for a great many disease agents, excretion with bodily wastes is a route to colonizing new hosts and even for completion of their life cycles. Although domestic animal wastes are not usually processed specifically for the destruction of disease organisms, practice has shown that they can be applied to field for growth of food crops without any apparent spread of disease to man or animals. Reservations to this generalization will be discussed later. Raw human wastes, however, used as fertilizer have been implicated in the spread of human disease, and processing before their use on crops certainly seems in order. Yet, processing as practiced in most modern sewage plants does not render either the sewage sludges or the sewage effluent produced completely free of pathogenic organisms. Therefore, the use of sewage sludge for soil amendment and sewage effluent for irrigation of crops must be approached with a certain amount of caution.

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Pathogens Associated with Biologic Wastes

The agents of disease that may be found in the fecal material and in some cases, in the urine of humans and other animals, can be classed into four groups of organisms: the viruses, the bacteria, the protozoans, and the helminths (worms). The intestinal worms, at least in their adult stages can be easily seen with the unaided eye. Protozoans and bacteria are microscopic while an electron microscope is needed to visualize viruses.

The enteric viruses infecting human beings include poliovirus, coxsackievirus (group A and B), echovirus, reovirus, adenovirus and the virus of infectious hepatitis. Poliovirus is, of course, the agent of poliomyelitis, a disease once greatly feared, but now essentially eliminated through mass immunization programs. Coxsackievirus, echovirus and reoviruses are usually associated with short-lived summer diseases, producing great discomfort including fever, chills, severe headache, vomiting, diarrhea and prostration. Adenovirus is associated with infection of the upper respiratory tract and conjunctivitis of the eyes. Of the above, the agent of infectious hepatitis is the only virus that so far has not been isolated and cultivated in the laboratory. Infectious hepatitis is primarily a disease of children and adolescents.

Viruses infecting animals other than man include those producing the avian leukosis complex and Newcastle disease in chickens; foot-and-mouth disease in cattle; scrapie in sheep; and transmissible gastroenteritis and hog cholera in swine. Of these, only the gastroenteritis virus is strictly an enteric virus, however, the others may become mixed with wastes

of their hosts and transmitted thereby. For instance, hog cholera is often spread by swine buyers, raisers, etc., carrying the virus in the manure on their shoes or boots from an infected hog lot to an uninfected one.

The most important disease producers among the enteric bacteria in the United States are those belonging to the genera Salmonella and Shigella. It has been estimated that there are around 2 million cases of salmonellosis (food poisoning) caused by salmonella bacteria in the United States each year. Two other diseases caused by organisms of the genus Salmonella, typhoid and paratyphoid fever, have almost been eliminated from this country, but occasionally, there are epidemics such as the recent typhoid outbreak in a Florida migrant labor camp. The prevalence of Shigellosis, or bacillary dysentery, caused by the shigella bacteria, appears to be related to lack of sanitation. The disease is relatively prevalent in the United States, but the form is not as virulent as that found in the Far East.

Both salmonellosis and shigellosis are zoonoses, that is, diseases that are transmitted among different species of animals, including man. For instance, salmonellosis infected chickens can transfer the disease via their eggs to humans.

The zoonoses, tuberculosis and leptospirosis, are diseases caused by bacteria found in fecal material. There are three kinds of tuberculosis-producing organisms. Those that produce it in man can infect, but not produce the progressive disease, in most other animals. Those producing the disease in cattle can infect man, but the disease is not expressed as human tuberculosis. The third kind produces the disease in poultry.

Human tuberculosis is no longer the most common cause of death in countries with a high standard of living, but it remains the number one killer of mankind in the world as a whole.

The medical and economic importance of leptospirosis is said to be underestimated in this country. This disease occurs throughout the world with the chief reservoir of infection the common rat. Dogs, cats, poultry, and a host of other animals, however, can become infected and transmit the infection to man.

Amebic dysentery of humans and coccidiosis of livestock and poultry are probably the only protozoan-caused diseases of importance in the United States. The ameobas causing these diseases cannot exist in their active forms outside the bodies of their hosts. A dormant form known as cysts of these organisms passes out in the hosts' excreta. The cysts can survive externally to their hosts and are capable of causing the disease if ingested. In man, besides dysentery, amoebiasis may produce liver abscesses. In livestock, coccidiosis is usually restricted to young calves and lambs. In poultry, older birds may be immune but productive young birds are often stricken with the disease. These organisms are host specific. For example, those causing the disease in sheep cannot do so in cattle.

The enteric worms, or helminths, infect the intestinal tracts of a great majority of the earth's animals and people. Most are zoonotic. Parasitologists class them as either flat or roundworms. The flatworms include flukes and tapeworms. The human flukeworm disease, Shistosomiasis, is not important in the continental United States, but is prevalent in Hawaii and Puerto Rico. Snails are alternate hosts in the disease cycle

for animals and humans. Animals are infected by ingesting cysts left clinging to leaves of grass by snails. Humans are infected in water by attack of a stage of the fluke that emerges from snails.

Infection with tapeworms (flatworms) is more common in the continental United States. Eggs of the tapeworm are excreted with human feces. If the eggs are ingested by cattle, these animals become infected with a form of the worm that burrows into the meat to form cysts. The cysts in incompletely cooked meat, are infectious to humans when consumed.

The roundworms or nematodes are probably more common in human and animal intestinal tracts than are the flatworms. It has been estimated that about 90% of the earth's human population is infected. Unlike the flatworms, most of the nematodes appear to be able to infect human beings without passage through an alternate host. Eggs of hookworms deposited with human fecal material in the soil, hatch and produce an immature form that infects by penetration of the skin of a person's bare feet or other bare areas coming in contact with the infected soil. Eggs of the roundworm, Ascaris lumbricoides, must be ingested before infection can occur.

Pathogen Survival in Soil

Ova of the intestinal worm, Ascaris lumbricoides, have been found to survive in a field soil up to 2.5 years. When exposed to desiccation on plant surfaces in a field, the survival time was less than 35 days. Cysts of the amebic dysentery producing protozoans, Entamoeba histolytica, have been shown to perish in soil within 8 days. When sprayed in conjunction with feces on plants growing in a field, investigators were unable

to recover infectious cysts after a period of 3 days.

There are pathogenic bacteria such as Clostridium tetani, the agent for tetanus and Bacillus anthracis, the agent for anthrax that produce spores. Spores are highly resistant to destruction. They may survive in soil for decades. Fortunately, the pathogenic bacteria encountered in sewage and animal manure, unlike the bacteria just mentioned, do not form spores. They live or grow best in their host. Some will grow in certain foodstuffs, or in special laboratory media, but for essentially all of these bacteria, the soil is an alien environment.

The bacteria of the genus Salmonella introduced into soil with sludge have been found to survive from 15 to 112 days. The human pathogen, Pseudomonas aeruginosa, applied to soil in sludge could not be recovered after 147 days. The erysipelas producing bacterium when experimentally inoculated into soil has survived no longer than 21 days. The tubercle bacillus, the agent of tuberculosis, has survived for 6 months in soil. In general, human bacterial pathogens that do not form spores may survive in soils for limited periods of time extending from a few days for some, to a number of months for a few.

Research on survival of bacterial pathogens in soil indicates that factors favoring the survival of the normal soil bacteria tend to favor the destruction of pathogens introduced into soil. Factors suppressing the growth of the indigenous microorganisms may conversely protect introduced pathogens. For instance, low moisture accompanied by cool temperatures have been shown to decrease the rate at which pathogens disappear from a soil. Beside inhibiting their predators or antagonists, part of the effect of cool temperatures may be, of course, to reduce

their normal rate of deterioration.

In order for viruses to reproduce themselves, they must be inside a living cell of their host. Viruses in the soil cannot increase in numbers. They can only survive until they come in contact with their host. Studies on virus survival in soil are very limited. Studies on their survival in natural waters, however, show that some viruses may persist from a few days to several months, depending upon pH, temperature, degree and kind of pollutants, and whether or not the water is in motion. If these survival times can be applied to soil, it would seem that the survival periods of viruses in soil under most conditions would be a matter of a few days, but could be extended to a few months under conditions of low temperature, moderate pH and medium to high moisture content.

Pathogen Movement in Soil and Water

The intestinal worms and their ova and the protozoans and their cysts are large enough that movement through a soil with water is very unlikely. Both the bacteria and particularly the viruses, however, are small enough that based upon size alone, movement through the pore systems of most soils could be expected.

At Pennsylvania State University, studies were made on the disposal of sewage effluent by sprinkling on land. Despite relatively high concentrations, at times, fecal coliforms in the effluent, the organisms were only rarely found to penetrate as deep as 2 and 4 feet.

Other field and laboratory studies have shown similar results. Even sands have been found to remove bacteria from percolating waters. The

mechanism of removal appears to be complex, involving straining, sedimentation and adsorption on particles.

Viruses, although much smaller than bacteria, also appear to be removed from waters percolating through soils. Work at Stanford University using bacterial viruses as models has been conducted utilizing nine California soils of varying characteristics. The bacterial viruses were readily removed from the percolating water, apparently by absorption, in even the coarse-textured soils.

One of the most reassuring studies with regard to removal of viruses from percolating groundwaters was conducted in California at the Santee Water Reclamation Project near San Diego. Attenuated poliovirus in high concentration was applied to the surface of a percolation pond used experimentally in the reclamation of sewage effluent for recreational use. The arrangement of the pond site was such that effluent from the pond was forced to flow underground 1,500 feet through sand and gravel overlying an impervious stratum to a collection ditch. Sampling wells were installed 200 to 400 feet down flow from the percolation pond. After application of the virus, sampling at the well sites was conducted periodically for 2 months. No poliovirus was found in any of the samples.

On the negative side, it must be realized that only a few experiments with either bacterial viruses or animal viruses have been conducted. Only a small sampling of the great variety of soils that exist have been used in these experiments. Much needs to be done to be able to predict just how long viruses will survive and move under a particular set of conditions in a soil profile. For instance, little is known about the effect of soluble organic materials emanating from sewage sludge on the

movement of viruses in soil. Some soluble organic materials are known to readily displace viruses adsorbed on clays and exchange resins.

Health Hazards in Waste Application

Danger to people applying chicken litter, animal manure, sewage sludge, and irrigating with sewage effluents can, perhaps, be evaluated by considering the health records of sewage plant workers. Surveys of sewage facilities in the United States have shown that there are for the most part, no special programs to immunize workers against the diseases to which they may be exposed. Exposure of sewage workers to disease probably occurs to the greatest extent via aerosols. Any imparting of mechanical energy to water such as running it through sprinkler nozzles, forcing air through it, or allowing it to flow over obstacles, will result in the production of very fine droplets of water that are capable of carrying viruses and bacteria into the air. When these droplets are inhaled, the size of the droplets will determine their penetration into the lungs. The relatively larger ones containing bacteria, are for the most part, captured by bronchial cilia and the bacteria that are removed from the inspired air may be swallowed. Smaller droplets carrying viruses may penetrate into the alveoli of the lungs. Based upon studies measuring the biological aerosols generated during aeration of raw sewage in the formation of activated sludge, it has been estimated that a man working within 5 feet of the downwind edge of an aeration tank would inhale a viable bacterial pathogen every two breaths. Yet, despite a lack of immunization programs and fairly intense exposure, it has been observed that sewage workers have fewer sick days than the general population.

Although this observation is comforting, the possibility of the increased morbidity rates due to work in sewage processing plants, cannot be evaluated on the basis of the workers alone. In addition, their families and their friends should be surveyed. Continuous low dosages of disease organisms may produce immunity and carrier status in the recipient individual and the result may be higher disease rates for people in his familial and social sphere. These kind of comprehensive studies as yet have not been made.

Besides the formation of aerosols, the production of dusts by dry materials such as poultry litter and the general contact involved in handling and applying solid and liquid wastes, certainly presents a disease hazard. The wearing of adequate clothing to cover as much skin surface as possible, and respirator masks when working with dry materials is certainly wise.

CONCLUSION

As previously discussed, it has been demonstrated that disease organisms can survive in soils and can contaminate foodstuffs grown upon soils. However, the missing link in incriminating soil as at least a temporary reservoir of disease is that infection depends not only upon the amount and the virulence of the agent that the victim happens to consume or contact, but upon the susceptibility of the potential victim.

There is experimental evidence that as little as one virus particle may transmit a disease and epidemiological evidence that very low levels of salmonellae in drinking water systems have infected large numbers of people. On the other hand, it is a reasonable assumption that we continually

consume a low level of salmonellae in our foods. Outbreaks of salmonellosis are mostly the result of the improper storage of food previous to consumption, allowing the organisms to multiply to infectious levels. Therefore, it is extremely difficult to evaluate the meaning of the presence of pathogenic organisms surviving in soil. It would be a mistake to conclude that because survival to a limited extent does occur, that all wastes should be incinerated. However, it would also be a mistake to become too casual and see no threat at all in the application of unprocessed human wastes to soils for the growth of crops used for direct human consumption.

The application of animal manures to land probably represents little threat to the health of either man or other animals with the exception of special situations. In general, the incorporation of manures by plowing into fields is a sound practice. The exceptions would be in the cases where the animals producing the waste are known to be infected with virulent quarantinable diseases such as anthrax, foot-and-mouth disease, and possibly, hog cholera. Applications of animal and poultry-house wastes to pastures may present special problems. For example, application of manure from animals with parasites directly to pastures would provide a very good opportunity for transmission of the parasites to uninfected animals currently grazing the pastures. Poultry-house wastes from a flock infected with salmonellosis would present a hazard to calves dropped by grazing cows, calves being particularly susceptible to the disease. Application of animal manure or poultry-house wastes to areas where runoff can contaminate ocean bays or estuaries used for the harvest of shellfish should be avoided. Shellfish are noted for their ability

to collect and concentrate viruses and bacteria from the water.

The application of human wastes is another matter. Several state boards of health have regulations governing their use. As an example, in 1951, the California Department of Public Health adopted regulations for the use of sewage in growing crops. The regulations prohibit the use of raw sewage on growing crops and provide that partially disinfected effluents shall not be used to water growing vegetables, garden truck, berries, or low-growing fruits, but may be used on nursery stock, cotton, and such field crops as hay, grain, rice, alfalfa, sugar beets, fodder corn, cowbeets and fodder carrots. In 1952, the Texas Board of Health adopted a resolution that no raw sewage could be used for irrigation regardless of the type of crop. The use of primary and preferably completely treated sewage was encouraged, however, for feed and pasture crops used for animal consumption. Sewage sludge was not recommended for use on crops to be used for human consumption.

Raw, untreated sewage, sewage effluent, sludge or septic tank pumpings, therefore, should not be considered for application to land being used for crop production. Under special circumstances, land may be used as a tertiary treatment for raw sewage, but this would be a non-agricultural use, and such land could only be returned to agriculture after disappearance of pathogens from the soil could be assured.

Anerobic digestion, composting, secondary activation, heating, drying, treating with lime, and aging of sludge, all contribute to the disappearance of pathogenic agents. Such treated sludges have been used

historically on land with no significant medical history of disease outbreaks. Even so, it is a wise precaution to discourage production of raw human food, especially vegetables, on land during the year of application of such treated sludges. After these "stabilized" sludge products have had a further year for decomposition in the soil, they should be safe for even raw vegetable production.

The above recommendations are admittedly conservative, but until much more information is available on the survival of pathogens, it is best to stay on the safe side.

Sludges stabilized as noted above are considered safe to use on all crops that are not used raw for human consumption. They can also be used on parks, roadways, golf courses and on lawns and landscape plantings without significant disease hazard. One should take note, however, of the odor problem which may be a nuisance around dwellings and of the limitations due to nitrogen and heavy metals mentioned in other sections of this report.

Additional Reading

Animal Diseases, The Yearbook of Agriculture, 1956, USDA, Wash. D.C.

An Evaluation of the Salmonella Problem, USDA, FDA and HEW. National Academy of Sciences, Wash. D.C. (1969)

Dunlop, S.G. 1968. Survival of pathogens and related disease hazards. In Municipal Sewage Effluents for Irrigations. Louisiana Tech. Alumni Foundation, Ruston, La.

Krone, R.B. 1969. The movement of disease producing-organisms through soil. In Municipal Sewage Effluents for Irrigations. Louisiana Tech. Alumni Foundation, Ruston, La.

Hanks, T. G. 1967. Solid Waste/Disease Relationships. A literature survey. USDA-HEW. Bureau of Disease Prevention and Environmental Control, National Center for Urban and Industrial Health. Solid Wastes Program, Cincinnati, Ohio.

Nitrogen Transformations in Organic Waste Applied to Land

Frank G. Viets, Jr. 1/

Organic waste includes animal manures and animal bedding, garbage, crop residues, sewage sludges, processing wastes from canneries, slaughter houses, pulp mills, and slash, sawdust and bark from the forest industry. Most of these materials have been used completely or to some extent in the past for growing crops, mulching, and erosion control. Agriculture, up until the end of the 19th century, was almost wholly dependent on animal manure and legumes to maintain the yield of crops. When available, manure was supplemented by sewage sludge, sewage after various degrees of treatment, raw sewage, and the blood and meat scraps (tankage) from abatoirs and fish canneries. Agriculture in much of the world, is still dependent on these sources of N due to the cost and lack of fertilizer nitrogen from the nitrogen-fixing industry. With organic N as the sole source of N, crop yields may be satisfactory if there is enough waste, but on the average, yields remain far below what the climate and good varieties are able to produce because there is not enough waste.

So, the use of organic wastes on land is nothing new, but their disposal and utilization has taken on some new aspects in these days of environmental concerns and modern ways of concentrating animals and people into small areas far removed from the land that produced their food. On the environmental side, two considerations dominate.

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First, organic waste must be kept out of water, as its biological oxidation (high biological oxygen demand or BOD) uses up the oxygen so that game fish die, the water becomes anaerobic, at least in the bottom, and stinks. Second, waste decomposition releases inorganic nitrogen that can support excess algal growth in water and nitrate that can cause methemoglobinemia in infants and various disorders in livestock.

With respect to concentrating people and animals in areas remote from where their food is grown, problems of insufficient land and high transportation costs back to the land have been created. Finally, the man who eats the food, or the cattle feeder or dairyman, often have no interest in the land that produced the feeds or food. Thus, organic wastes that were once regarded as a valuable asset in agriculture have often become a nuisance problem in which some of the alternatives have been to dump at sea, float down the river, or accumulate in huge mountains or sanitary landfills. Tendencies to use waste nonbeneficially in the past have been accentuated by low prices for fertilizer nitrogen, phosphorus, and potash which are modern alternatives to the use of manure and other organic wastes. Recent shortages of nitrogen fertilizers and higher prices have altered the situation.

Various State and Federal pollution control laws finally culminating in the 1972 Amendments to the Water Pollution Control Act (PL 92-500), that eventually demand zero release of pollutants, means that disposal of organic wastes on land cannot pollute surface or ground water. Other alternatives for use such as incineration, power production, and use as building materials are beyond the scope of this guideline.

As mentioned, organic wastes generally contain nitrogen in the form of ammonia, urea, amino acids, plant and microbial protein, that upon decomposition, yield inorganic nitrogen (ammonium or nitrate) that can lead to overfertility of surface water, and nitrate in surface or ground water that can harm people and animals. There are many other sources that can contribute nitrogen to surface and ground waters, including rain, fertilizers, cesspools, septic tanks, sewage plants, and natural deposits of nitrate, but these sources are beyond the scope of these discussions relating to land application of organic wastes. Likewise, much controversy exists on the levels of nitrogen needed in water for eutrophication; whether there are any controls that will maintain sufficiently low levels considering the multitude of sources that existed in nature before the advent of man, and whether too much emphasis is being placed on control of nitrate in water supplies. These points are mentioned to point out that the whole burden does not rest on controlled disposal of organic wastes on land.

Concepts of Waste Disposal on Land

Two concepts of organic waste disposal on land are important for they affect the management and the ultimate fate of N in the environment. The first concept is that wastes should be used as a resource for their nutrient content and favorable effects on soil tilth for the production of crops and forages in sustained management systems. This concept involves the use of higher loading rates than were generally used heretofore when wastes were less abundant. It involves use in ways that maximize N availability to plants, and minimize loss of N by denitrification, runoff of

organic N, and deep percolation of nitrate. Hopefully, the costs of application will be more than offset by the increased returns on crops.

The other concept is that land can be used simply as a disposal site in which crop production is only of incidental importance or is not practiced at all. The sanitary landfill is such a disposal site, but these guidelines are focused on incorporating wastes into soil rather than simply covering up a great compacted depth of them with soil. In land disposal, the emphasis is on maximizing the decomposition rate and loss of N by denitrification, but again, minimizing N loss to surface and subsurface water. In such systems, cost of waste disposal must be paid or passed on some way to those who created the waste. There are no beneficial effects other than simply getting rid of something unwanted.

Nitrogen Transformation in Soil

All organic matter disposed of on land immediately begins to be decomposed by microbes in the soil or in the waste, provided temperatures are above freezing and sufficient water is present. Even under low temperatures and drought, some decomposition occurs due to the release of heat and moisture by the decomposing organisms. Some of the ammonia and nitrate as well as soluble nitrogen compounds will diffuse out of the decaying waste. Additional amounts of these compounds are released as decomposition proceeds.

Several important conditions in the waste and in the soil environment determine what and how fast they will be released. These are important parameters in the nitrogen cycle, as indicated in Figure 1. The nature of the end products depends on the presence or absence of oxygen during decomposition. In the presence of oxygen (aerobic conditions) the end

products are humus-like, nitrogen-containing products that become increasingly resistant to oxidation as decomposition progresses, and ammonia which can escape into the air or be nitrified to nitrite and then to nitrate. In the absence of oxygen (anaerobic conditions), the end products are humus-like materials, ammonia, and nitrogen gas. The latter is harmless in the environment as air is about 80% nitrogen gas. Some oxides of nitrogen other than nitrate may be produced, but the importance of soils as a source of them is probably unimportant. Some evidence is accumulating that denitrification may occur even though a part of the air-filled pore space of the soil contains oxygen. Oxygen-deficient or anaerobic conditions may exist in and around chunks of decomposing waste and in the pore space of soil aggregates.

Several conditions determine whether a given site of microbial activity in a soil will have free oxygen present. These include the water content of the soil, its air-filled pore space, and the biological oxygen demand or rate of oxygen use. Conditions that favor high rates of oxygen use are high temperatures promoting rapid oxidation, high water content, amounts of waste added per volume of soil, and the waste's decomposability. Thus, the factors affecting organic decomposition rates are tied closely to the rates and kinds of nitrogen products, with the oxygen supply being a primary determinant. The conditions of decomposition may be entirely different for a loading rate of 100 - 200 tons of manure per acre than for a low rate of 10 tons per acre or less on which most of our literature is based and is common in farm practice. Likewise, the aeration and decomposition conditions may be quite different for 10 tons of sewage sludge incorporated into the surface soil than they are for 200 tons per acre plowed down in a strip at a depth of 2 feet.

In spite of the many investigations of denitrification that have been conducted in laboratories to explain the mechanisms and the conditions for denitrification, practically no knowledge is available from field situations on how much of the nitrogen in a waste is lost by denitrification. The reason for this lack of information is that the product, nitrogen gas, is identical with that in the soil and atmosphere above it. The only tracer of N has been the heavy isotope ^{15}N that is too expensive for use in field studies.

Another important characteristic of nitrogen in waste is the amount and kind of carbon compounds that are associated with it. The carbon-to-nitrogen ratio or percentage of N tells part of the story, but not the whole one. The carbon compounds in waste plastics may be extremely resistant to decomposition so that the associated nitrogen stays locked up. In the commercial manufacture of fertilizer materials called urea-formaldehydes, too many cross links in the polymers led to products with no fertilizer value. Wastes that contain a high proportion of lignin in their total carbon such as wood chips and sawdust, or wastes that have a high proportion of keratinized protein such as chicken feathers and animal hair, do not behave in accordance with the usual simple carbon-nitrogen ratios.

If we eliminate some of the special situations mentioned above in which the organic matter is extremely resistant to decomposition and confine the discussion to the usual kinds of plant residues and manures, then the carbon-nitrogen ratio becomes of considerable importance in predicting how the addition of waste is going to affect crop growth and the amount of nitrate available for leaching. When organic matter

is added to a soil and temperature and moisture are favorable, decomposition, by fungi, actinomycetes, and bacteria progresses very rapidly. Carbon is lost as CO_2 or incorporated into organisms as structural components. If the material has a very high carbon-nitrogen ratio (low percentage nitrogen), all of the N released by decomposition is re-incorporated into organic N compounds presumed to be largely proteins. This is called N immobilization. Carbon content of the residues rapidly decreases and organic N content increases. As decomposition proceeds at a progressively slower pace, the carbon-nitrogen ratio narrows until finally nitrogen begins to be released in inorganic form and is usable by growing plants. If there are no growing plants, the end product, nitrate, can be leached.

In practice, the net result of use of high carbon-to-nitrogen ratio material may be depressed crop yields when the soil lacks sufficient available nitrogen at the start. This situation is frequently encountered when cereal straws or corn stalks are incorporated in a soil and crops are planted within 1 or 2 weeks. Manure containing large amounts of straw or wood products as animal bedding can produce the same effect. The greater the time available for decomposition before a crop is planted, the less the yield-depressing effect. Materials containing less than about 1.3 to 1.4% on a dry weight basis will depress yields of non-legumes planted soon thereafter. The quantity of material added is important too. Thus, a cereal straw containing only 0.5% N with an N deficit of 0.8% (1.3-0.5%) would need about 16 pounds of fertilizer N per ton just to offset the N depressing effects of the straw. Exact numbers cannot be

given, as time and conditions affecting rate of decomposition vary. However, the numbers given are a useful guide.

Nitrogen-rich plant residues such as legume cover crops, young grasses, and animal manures without bedding materials may have such low ratios of carbon to nitrogen that they begin to release nitrogen in inorganic form very soon after addition to soil. This nitrogen can be used by crops, or as nitrate, can be leached. A rule of thumb from the literature is that a residue must contain about 1.8% N before it will release significant amounts of nitrogen.

Since only a part of the carbon in organic residues is oxidized in the soil, the residues becoming progressively less oxidizable with time, these resistant residues become a part of the humus of the soil. Associated with and part of the humus, are nitrogen compounds that are resistant to further mineralization and oxidation. Because of this resistance to decomposition, at most only about half of the total N in organic residues can be regarded as available for plant growth within 2 to 3 years after addition to the soil. In the first year the N released may be negligible. When organic wastes are added to soils in tremendous quantities of 100 dry-weight tons or more per acre, the nitrogen available for crops may be much less or even none at all because of denitrification induced by high BOD of the decomposing residues. For denitrification to occur there must be some zone in the soil sufficiently aerated for nitrification to produce nitrate. This nitrate can then be moved into a poorly aerated zone where denitrification can occur.

Much of the experience with nitrogen transformations in soils has been done at loading rates common in farm practice, and much less is known about aeration, possible toxins, and salt effects of massive loading rates.

Importance of the Amount and Timing of Water Available

Water is a major consideration in designing systems of waste disposal, whether the system be for use in growing of crops or simply getting rid of waste. Water is essential for decomposition, the transport of soluble or suspended organic N from the site by overland flow, and deep percolation of nitrate. Water content also partially determines the extent of denitrification. Although the role of water appears self evident, there are many examples of failure to consider it. Great hazard to surface water can be created when organic wastes such as manure are spread on soil with high runoff potential, such as fine-textured soils on slopes, areas immediately adjacent to lakes and streams, and on frozen soils. The same soil conservation principles applied to keep soil in place must be applied to keeping organic wastes in place. Most organic wastes have low densities and wet up slowly so that they readily float away. For these reasons, wastes should not be spread on frozen soils, applied near streams, nor be left on the surface without incorporation where rainfall intensity is high. Based on studies of waste water renovation by overland flow, organic wastes should not be left on the soil surface within 175 feet of a stream. Such a rule is generally not difficult to follow as soils along streams are usually fertile and do not need the nutrients in organic wastes.

In semiarid regions, special thought must be given to using N-rich wastes such as manures in cropping systems. Water is more often the factor limiting crop yield than the N supply. Use of heavy applications of manure or sewage sludge can stimulate vegetative growth of dryland

crops in the early summer to the point that stored soil water is exhausted and the crop suffers drought. Crop failure can result. Large amounts of organic waste can be used beneficially only where adequate water is available from rainfall or irrigation. On the other hand, semiarid areas--the drier the better--offer opportunities for simple disposal of wastes by soil incorporation. Because of shortage of precipitation, little leaching of nitrate can occur. Such disposal systems should be managed with a cover crop to exhaust the moisture stored in the soil so that the probability of leaching can be further reduced. A condition resembling summer fallow should be avoided. Unfortunately, semiarid areas are usually too far away from cities and feedlots where waste disposal is a problem to be of much significance for waste disposal. Exceptions occur near high density metropolitan areas in California, Arizona, Utah and Colorado.

Use of Wastes in Crop Production Systems--Low N-High Carbon Wastes (C:N Ratio About 35)

These materials include mature cereal straw, timber waste, vegetable cannery wastes, and some manure that contain bedding and are highly weathered. These materials usually contain only 0.5 to 1% N. These materials contain so little N and so much oxidizable carbon that they can accentuate nitrogen deficiency. The magnitude of the effect depends on the amount applied, its N content, and the time allowed for decomposition before a crop is planted. So many variables are involved that laying down definite guidelines is difficult. We suggest that for residues incorporated into the soil before a crop is seeded that the usual N application rate be increased 1 pound per acre for each 0.1% N the residue is below 1.3% N per thousand pounds dry weight of residue.

For 6,000 lbs of 0.5% N straw, this amounts to 48 lbs of N $[6 \times (1.3 - .5\%)]$. This rule applies to wastes brought to the site and not to the plant residues usually produced on the site. Application of this additional N can be avoided if N-fixing clovers, alfalfa, or soybeans are crops that follow the heavy residue applications. Perhaps, use of legumes where adapted following incorporation of large amounts of low N residues, is the best way to utilize such residues.

Disposal of on-farm produced residues such as straw and stover is usually not a problem. Cannery wastes can be. Disposal of forestry waste products is becoming less of a problem because of their more complete utilization in chip board and other industrial products of the integrated lumber industry.

Much has been written about use of low-N residues to increase soil organic matter, to rebuild the soil to the organic content it had when first plowed and the use of N fertilizer to "stabilize" carbon residues in the soil. The results of experiments to use manure up to amounts of 20 tons per acre-per year in increasing or even maintaining soil organic matter have mostly been disappointing to those with that goal in mind.

The addition of N fertilizer to organic wastes low in nitrogen can offset the N-reducing availability effects of the low N residues, will hasten the decomposition of the residues, but will not result in "stabilization" of more organic matter in the soil.

Since so many local soil variables and management variables enter into recommendations on how best to utilize fertilizer N in conjunction with low-N wastes, broad guidelines are of little assistance.

High N (low C:N ratio) wastes (C:N ratio 25 or less)

Wastes with a high N content include fresh animal feces, sewage sludge, some (but not all) legume straws, and the residues of most all young plants clipped or plowed under in vegetative stage. Some crop residues once regarded as low in N, such as corn stover, may now have sufficient N because of ample N fertilization to be regarded as high N residues. Sewage sludges vary considerably in composition depending on the digestion processes used. Digested sewage sludges may contain about 5.6% N on a dry weight basis.

The value of animal manure as a nitrogen source for crops is highly variable. This variability is to be expected in view of the enormous variations in N content resulting from differences in kind of animal, their diets, and the system of collection, storage, and handling of the manure. Fresh animal feces (solids plus liquid) vary in nitrogen content from about 2% for hog waste to 4.0% for sheep manure when expressed on a dry weight basis. However, at least half of the N is in the urine as urea (uric acid mostly for poultry) which quickly hydrolyzes to ammonia. Much of this ammonia escapes into the air on storage and partial drying. Preservation of the N content of manure has always been a problem which, in the past, led to composting with superphosphate to neutralize the ammonia. The conditions of collection influencing the retention of urine, the amount of drying leading to loss of ammonia and denitrification in the manure pile, and the exposure to weather after spreading, are only a few of the conditions that influence the quality of manure as actually used. The amount and kind of bedding

is another variable affecting nitrogen availability. All of these factors have led to very variable response to manure. In some cases, it has depressed yields. In other cases, it has been highly favorable for crop growth provided sufficient moisture was available.

In general, high N wastes will release some N to immediately following crops if the N content on a dry weight basis is above 1.5%, a C:N ratio of about 25. In order for a waste to yield significant amounts of N, the N content must be 2% or greater. At most, an organic waste might be expected to release in available form for a subsequent crop about 1 pound of N per 1,000 pounds dry weight for each 0.1% N over 1.5%. Thus, a ton of dry manure with an N composition of 2.0% might yield at most 10 pounds of available N $[2,000 \times (2.0 - 1.5\% \text{ N})]$. a 120-bushel-per-acre corn crop with a total N requirement of 190 pounds of N would require 19 tons of dry manure with an N composition of 2% N to supply all of the N needs. This would leave 570 pounds of N (760-190) per acre from the manure still in the soil, a portion of which would become available in subsequent years. A portion of this 570 pounds of N might be denitrified and some of the N would enter into the decomposition-resistant pool of humus in the soil.

Some precautions must be taken on rates of application of N-rich wastes when they are used on pastures for livestock. In the South, large amounts of poultry litter applied to pastures in fescue have led to grass tetany, "fescue toxicity," fat necrosis, and lack of milk production and the possibility of nitrate toxicity in cattle. Four

tons per acre of poultry litter applied annually appear to be a safe rate. Since grass tetany is a widespread malady in cattle grazed on fast-growing herbage usually induced by high N and potash, and low magnesium, precautions should be taken everywhere large amounts of N-rich wastes are used on pastures.

Loading of Land for Waste Disposal

Under this concept land is loaded to its limit with waste, with the limit for N loading being that nitrate not escape into runoff or percolating water. Methods of application of the waste may be incorporated into the surface soil or at considerable depth with special equipment such as deep plows, trenchers, or land filling. The rates of application may be in the range of 20 to several thousand tons per acre. Decomposition rates per unit of waste are undoubtedly slower than under the usual conditions of agricultural use because of the high concentrations per unit of soil or the deliberate concentration in a trench in which the contact with the soil is reduced. Conditions for denitrification may be greater. Ammonia production in the immediate vicinity of decomposition sites may inhibit nitrification so that ammonia toxicity to a part of the decomposing microbes results. Overall, conditions for water pollution with nitrate per unit of waste applied would appear to be less under these systems than in agricultural systems where the objective is to grow as much crop and dispose of as much waste as possible in concurrent operations.

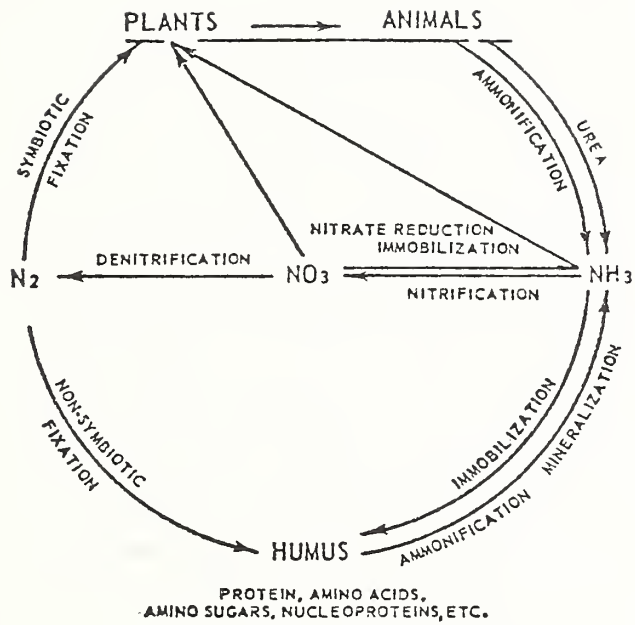


Fig. 1-- The Nitrogen Cycle

SOIL ADSORPTION OF PHOSPHATES
FROM WASTE WATER

A. W. Taylor and H. M. Kunishi^{1/}

Waste water can be applied to the soil by three principal techniques, overhead irrigation, surface irrigation on cropped soil (i.e., overland flow) or impoundment for vertical percolation through the soil. Although in the first two some of the applied water may be used by the evapotranspiration of the crop so that the amount of water reaching the water table or being discharged as sub-surface drainage will be controlled by the balance between the application rate and the crop consumption, some degree of water movement through the soil takes place in all three cases.

The purpose of this chapter is to describe the basis of an empirical method to estimate the amount of phosphate that will be removed by a soil from a phosphate-rich water percolating through it. In effect, this reduces to a measurement of the phosphate adsorption isotherm. Owing to the complex character of the reactions by which phosphate is adsorbed by or precipitated on the soil colloids, some appreciation of the general character of phosphate chemistry in soil is essential if the measured isotherms are to be usefully employed in predicting the behavior of field systems.

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The first section will therefore be devoted to a discussion of the points that must be borne in mind in the development and use of the proposed procedures.

The forms in which phosphate is adsorbed on soils include basic aluminum, iron and calcium phosphates. The aluminum and iron compounds are of indefinite chemical composition and may exist as surface coatings on oxides or hydroxides of these metals in the soil. Calcium phosphates, which only occur in soils above pH 8, are related to the apatites. In soils of high phosphate status octacalcium phosphate may be present at such pH values: this is one of the few discrete mineral forms that have been identified. In the neighborhood of dissolving fertilizer granules the presence of taranakites (a complex aluminum phosphate) and dicalcium phosphate may sometimes be inferred, but such minerals are unstable in the general soil environment. They are unlikely to form from phosphate added in waste materials and need not be considered in the present context.

A highly simplified mechanistic picture of the way in which phosphate adsorption takes place is presented in figure 1a. Under quasi-equilibrium conditions a steady state may be considered to exist between the phosphate in solution and that present in the more readily available forms adsorbed on the soil. When the phosphate concentration is increased, adsorption reactions lower the concentration to a new value as increasing amounts are adsorbed. If the phosphate solution

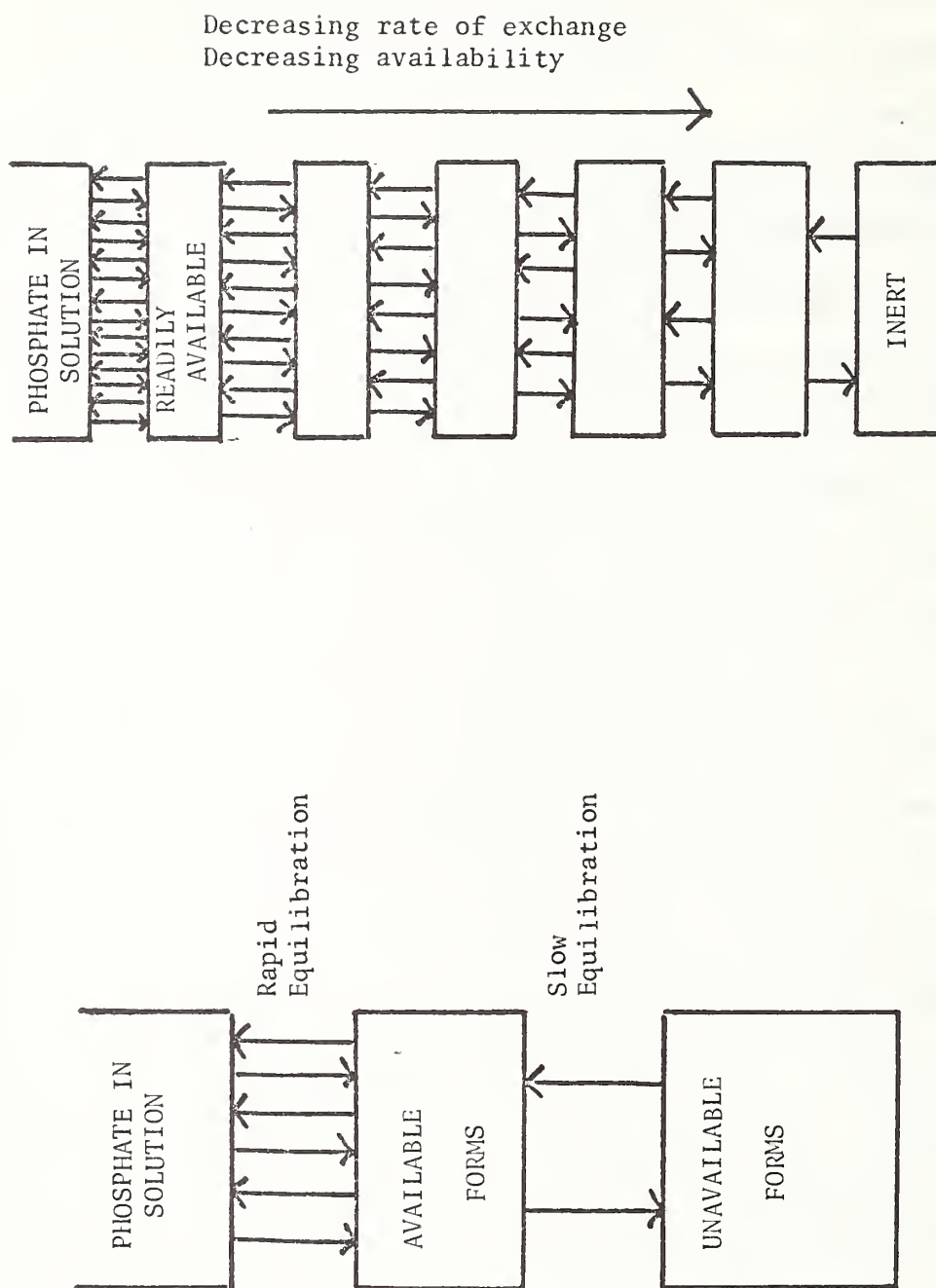


Figure 1a

Figure 1b

Figure 1. Conceptual models of the mechanism of phosphate adsorption in soils.

is replaced by water, desorption will take place at the expense of the available reserve.

The third compartment in figure 1a representing "fixed" phosphate comprises more insoluble and less reactive forms. Equilibration between these and the more readily available phosphate takes place over long periods of time. The phosphate in these unreactive forms is not available as a source of this nutrient for plants and does not readily dissolve in the soil water. The actual chemical compounds in this category are ill-defined. In acid and neutral soils they are probably mainly basic iron phosphates of variable composition. In calcareous soils apatite-like materials are also likely to be present. The unreactive nature of these compounds, which comprise the bulk of the total phosphate present, is the reason why measurements of the total phosphate content do not give any useful measure of the phosphate fixing capacity of a soil. This is dominated by the more rapid adsorption reactions of the reactive fraction.

The picture delineated in figure 1a is however too simple to provide a useful basis for the interpretation of many known aspects of phosphate behavior. In reality there appears to be a continuous spectrum of adsorption states ranging from a fraction in immediate equilibrium with the solution, through more strongly adsorbed and less labile forms to the inert fraction, as illustrated diagrammatically in figure 1b. While the concentration of phosphate in the solution is governed by the adsorption and desorption reactions of the most readily

available fraction, any changes in the amount of this will ultimately be reflected by slow changes in the amounts of the less readily available forms. The immediate equilibria with the solution are usually established in a period of hours, but the readjustments in the more strongly bound forms may take place over months or years. Adsorption reactions are therefore not readily reversible and hysteresis effects are commonly observed in successive adsorption and desorption experiments. Similar effects are also found when the solution phosphate is labeled with the radioactive ^{32}P isotope. The size of the phosphate pool in the soil that this dilutes into is invariably found to be larger than the amount adsorbed over short time periods, and not all of the labeled material can be reversibly recovered.

Although the chemical character of the adsorbed phosphate pool is very complex, a full understanding of the system is not necessary to assess the capacity of a particular soil for removing phosphate from water. This can be done on the basis of determination of the Adsorption Isotherm, which describes the relationship between the phosphate concentration in the solution as a function of the amount that has been adsorbed by the solid phase.

Experimental Determination of Adsorption Isotherms

If a series of 5 g samples of a soil are shaken with 100 ml volumes phosphate solutions of known initial concentration, ranging from 5 to 50 ppm of P,* and the solutions are then centrifuged and analyzed, adsorption isotherms similar to those in figure 2 may be constructed. In these graphs the final phosphate concentrations are plotted on the abscissa, and the ordinate represents the amounts of P taken up by the soil. These curves describe the phosphate concentrations that the soil will support in the solution after the adsorption of different amounts of phosphate. Thus, soil A, which has a relatively high adsorption capacity, will not maintain 10 ppm in solution until it has adsorbed 970 ppm of its own weight. In contrast, soil D, a low adsorber, will be in equilibrium at this concentration after taking up 130 ppm. The slope of the curves decreases as more phosphate is adsorbed, suggesting that there may be a limiting saturation capacity when very large amounts are added. This can however rarely be experimentally measured. Many attempts have been made to interpret the shapes of isotherms in mathematical terms using the Freundlich or Langmuir models, but it is difficult to interpret these in terms of the actual adsorption process.

* While phosphorus is invariably present as one of the ions of phosphoric acid, H_2PO_4^- , $\text{HPO}_4^{=}$ and $\text{PO}_4^{=-}$, it is convenient to reckon solution concentrations and adsorbed amounts in terms of elemental phosphorus. For solutions, parts per million and $\mu\text{g P/ml}$ are numerically identical. Quantities adsorbed are most conveniently measured in ppm (micrograms/gram) of oven-dry soil.

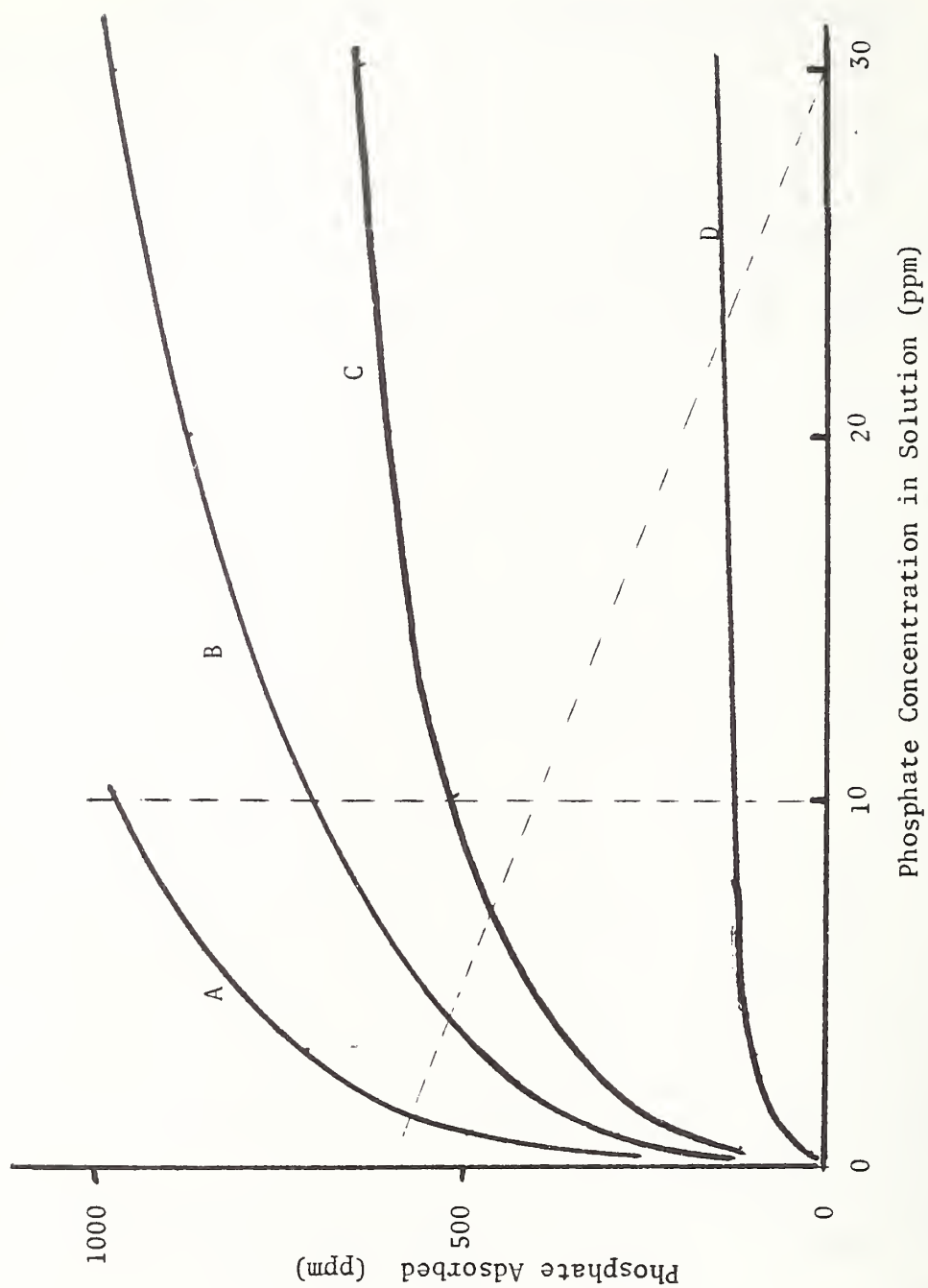


Figure 2. Adsorption Isotherm of four soils plotted in linear coordinates

The Freundlich isotherm is described by the general equation $x = Ac^{1/n}$, where x is the quantity of phosphate adsorbed when the solution concentration is equal to c , and A and n are constants. This equation implies that the energy of adsorption decreases exponentially as the adsorption increases. The validity of this assumption may be tested by plotting the data according to the logarithmic form of the equation $\log x = \log a + (1/n) \log c$, when a straight line should be obtained. Soil phosphate data usually give gentle curves, indicating that the implicit assumption that the energy of adsorption decreases exponentially with increasing adsorption is not valid except as a very loose first approximation. The Freundlich equation does not imply the existence of an adsorption maximum.

The Langmuir equation is based on the assumption that the energy of adsorption is independent of the number of adsorption sites filled, but that there are a limited number of these. The equation is written as $c/x = c/x_m + 1/Kx_m$ where x_m is the maximum adsorption and K is a constant. If a straight line is obtained when c/x is plotted against c , the slope defines the adsorption maximum in terms of $1/x_m$. Soil phosphate isotherms are again usually curved so that maximum adsorption capacities cannot be defined with precision. The reason is that neither of the basic assumptions of the Langmuir treatment are valid. The existence of a spectrum of adsorption substrates, as illustrated in figure 1b, implies that there must be a number of sites with different adsorption energies available to the solution ions, and

the behavior of these may depend upon the number occupied. The latter clearly happens where the adsorption is due to the formation of surface layers of basic iron or aluminum phosphate upon oxide or hydroxide particles. The composition and solubility of these coatings may vary with the composition of the solution with which they are in contact.

Under these circumstances the most convenient form of the equation is a modified form of the Langmuir,

$$x/x_n = (RT/b) \log Ac,$$

where A and b are constants and x_n is the hypothetical maximum adsorption capacity. If x_n is treated as another constant, the equation becomes $x = K \log Ac$ where K is a complex constant. The development of this equation contains the assumption that the energy of adsorption decreases linearly as adsorption increases. Since this assumption is not met, a plot of x against $\log c$ cannot be expected to be a perfect straight line, but data from many soils give isotherms that are approximately linear over limited ranges of concentration. The equation therefore provides an approximation that can be invoked to justify the use of a logarithmic concentration scale in order to simplify the graphical treatment of the isotherms.

Figure 3 presents the data of figure 2 transformed in this way. In addition to giving straighter curves, the logarithmic scale reveals the shapes of the isotherms at low adsorption levels in more detail. The compression of the data on the linear plot of figure 2 conceals the fact that the isotherms for soils B and C cross at a

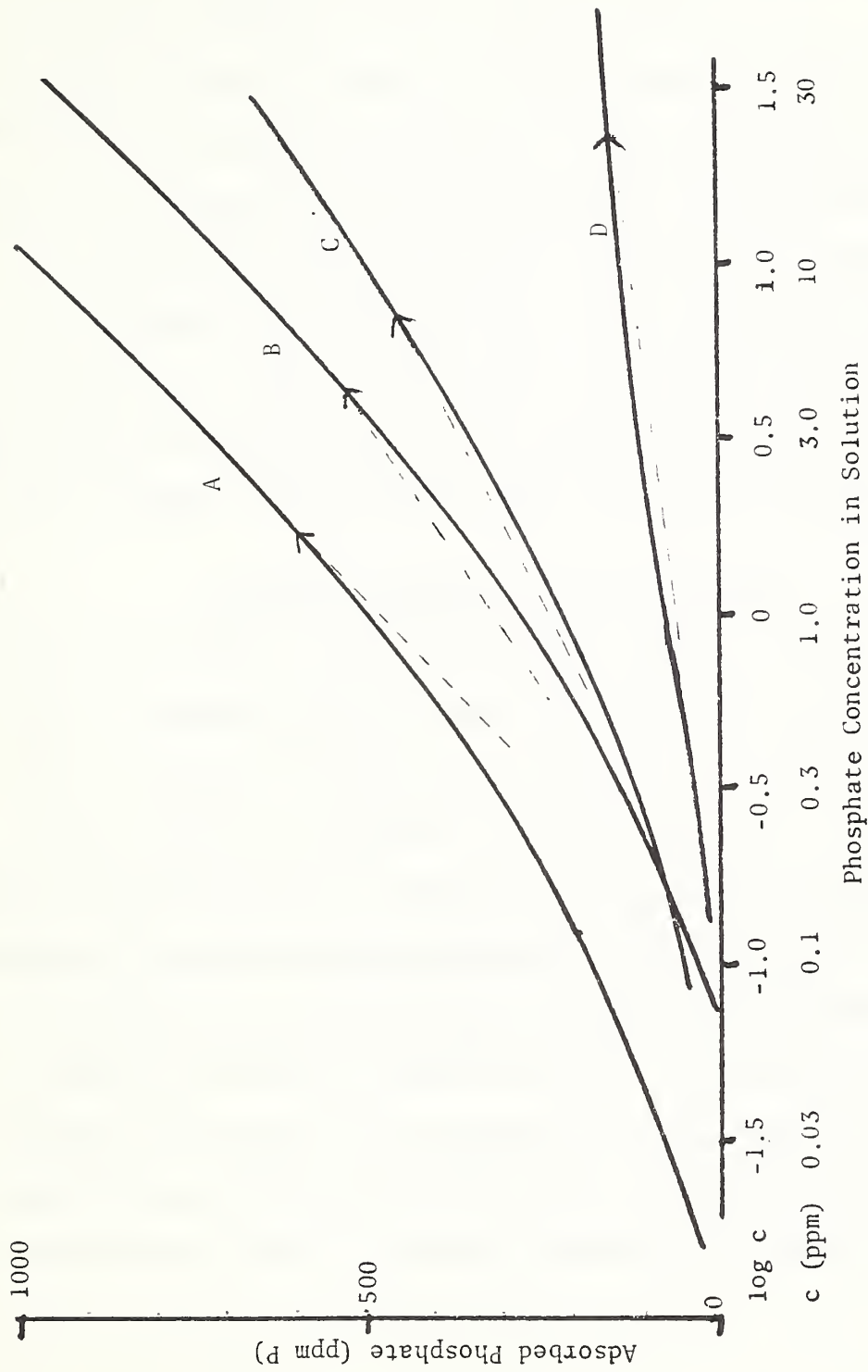


Figure 3. Adsorption Isotherms of four soils plotted in logarithmic coordinates.

solution concentration of 180 ppb. This arises because the phosphate concentration in solution in equilibrium with samples to which no phosphate has been added is higher in soil B (about 60 ppb) than in soil C (20 ppb) although soil B has a higher adsorption capacity at large phosphate additions. Changes in the natural equilibrium concentrations may result from the addition of phosphate fertilizers. Application of 100 lbs/acre of P to a 4" depth represents the adsorption of about 100 ppm: this is obviously too little to cause changes in all but the lowest part of the isotherm. The importance of measurements in this concentration range for the detailed prediction of the behavior of individual soil profiles will be considered below.

Interpretation of Adsorption Isotherms

In order to estimate the absorbing power of a soil for phosphate it is necessary to determine the amount of phosphate that the soil must adsorb in order to support a given phosphate concentration in the water. This is equivalent to a determination of the slope of the adsorption isotherm over the relevant solution concentration range. Where the full adsorption isotherm has been established, this may be done graphically by measuring the amount adsorbed at an arbitrary chosen concentration. If this is 10 ppm, the slopes of the four isotherms in figure 2 can be related to the indices of 970, 690, 505 and 130 ppm of adsorbed P. This index however gives no information about the shape of the isotherm: it does not reveal that adsorption

in soil A is almost continuous up to the 10 ppm concentration, whereas soil D has almost reached saturation capacity at 4 ppm.

The method also has the great disadvantage that four or five measurements must be taken for each soil, greatly increasing the work needed when a number of soils are to be screened. If the condition that comparisons must be made at the same concentration is discarded, then single measurement determinations can be made by comparing the amounts of phosphate adsorbed from phosphate solutions of the same initial phosphate concentration. If this is chosen as 30 ppm, the isotherms of figure 2 could then be characterized by the values of x/c --

	<u>x</u>	<u>c</u>	<u>x/c</u>
A	566	1.7	333
B	518	4.1	126
C	458	7.1	64.5
D	144	22.8	6.3

While this method has the advantage of requiring only a single measurement, it gives no direct measure of the relative adsorption over any specified concentration range. The comments made concerning the inadequacy of the 10 ppm intercept apply with even greater force. The intercept on isotherm D at 22.8 ppm P is an even poorer index of the adsorption characteristics of this soil than the 10 ppm value, and the value of 333 for soil A, measured at a solution concentration of 1.7 ppm is a serious underestimate of the adsorption capacity in the working range where solutions may contain between 5 and 10 ppm P.

A more useful form of a single-measurement index may be developed by taking advantage of the logarithmic form of the isotherms plotted in figure 3. The slope of this curve between two values of x is given by $(x_2 - x_1)/(\log c_2 - \log c_1)$ where c_2 and c_1 are the corresponding concentrations. Since the slope of the logarithmic isotherm is relatively constant, it can be derived from a single measurement provided that the experimental conditions are within the range of practical interest between solution concentrations of 1 and 10 ppm. This condition will usually be met when a solution containing 0.6 mg of P per gram of soil is used to make a suspension with a soil: water ratio of 1:20. For 5 grams of soil in 100 ml of solution this corresponds to an initial solution concentration of 30 ppm: the phosphate may be conveniently added in the form of KH_2PO_4 . The solution should also be 0.02 M or 0.13% in KCl: the latter salt is added to keep the system flocculated to facilitate filtration or centrifugation for analysis. The final phosphate concentration, c , is measured after shaking for 12 - 16 hours and the value of x , the amount adsorbed, is calculated from the change in phosphate concentration.

The slope of the isotherm is then calculated in terms of the ratio $x/(1.0 + \log c)$. The use of the term $(1.0 + \log c)$ compares the slopes of the isotherms on the assumption that all have intercepts with the abscissa ($x_1 = 0$) at $\log c_1 = -1.0$. This is equivalent to assuming that all the soils contain a native phosphate solution of 100 ppb of P. This will be rarely found to be the case, but it

provides a sufficiently good uniform point of reference for rapid comparisons of the adsorption capacities in the 1 to 10 ppm range derived from simple one-solution measurements. In actual terms the ratio $x/(1.0 + \log c)$ gives an estimate of the average amount of phosphate (in ppm soil) that will be adsorbed for each tenfold change in solution concentration between 0.1 and 10 ppm. The extent to which this estimate corresponds to the actual behavior of the four soils of figures 2 and 3 may be seen from the following table.

<u>Phosphate Adsorption</u>				
	<u>0.1 to 1.0 ppm</u>	<u>1.0 to 10 ppm</u>	<u>Average</u>	<u>$x/(1.0 + \log c)$</u>
A	310	480	395	460
B	290	400	345	322
C	165	275	282	246
D	65	65	65	61

The overestimate for soil A is due to the failure of the assumption that the native phosphate concentration is close to 100 ppb in this soil. The true value is closer to 10 ppb, and the soil has an adsorption capacity of 180 ppm up to the 100 ppb concentration level.

Since this procedure only requires a single measurement it provides a suitable method for rapidly determining the relative capacities of a number of soils. It may also be applied to measuring differences in adsorption capacity with depth in a profile where information is required about the capacity of a soil to remove phosphate from water percolating through it. Such profile surveys are always desirable since differences in texture and mineralogical character may cause considerable changes in adsorption capacity with depth.

Predictions of Lower Concentration Limits

While the adsorption isotherms discussed in the preceding section are appropriate for estimating adsorption from concentrations of 2 to 5 ppm that may be encountered in sewage effluent and waste-waters, they give little information about the lowest levels to which the concentration can be reduced as it percolates down the profile. In order to estimate this it is necessary to examine the lower range of the isotherms in more detail.

In normal soils the native pool of adsorbed phosphate is sufficient to support concentrations in the range from 5 to 250 ppb, depending upon the amount of fertilizer the soil has previously received. Such fertilizer phosphate is usually confined to the surface horizons, and the concentration in subsoils is usually very low. As soon as waste-water is applied the phosphate status of the topsoil will increase rapidly and water of steadily decreasing concentration will move into the lower horizons. The ultimate limit will therefore be determined

by the adsorption isotherm of the subsoil. This must be determined in a separate set of experiments using solutions with phosphorus concentrations in the 50 - 500 $\mu\text{g/ml}$ range.

These measurements are made in a similar manner to those described above, except that at least three equilibrations are necessary. In this procedure three 5 gram samples of soil are shaken for 4 hours with 100 ml volumes of solution containing 200, 100 $\mu\text{g P/ml}$ and zero phosphate. The solutions should all contain 0.13% of KCl as before. After centrifugation and analysis of the final equilibrium solution the amounts of phosphorus adsorbed (or desorbed) are then plotted as a function of the final phosphorus concentration as illustrated in figure 4a. Since the third solution contains no added phosphorus, the amount released to the solution will represent desorbed phosphate and must be plotted below the abscissa. The intercept of the isotherm with this axis gives an estimate of the phosphate concentration that is in equilibrium with the native adsorbed phosphate on the soil. This figure is an index of the lowest phosphate concentration that can be expected in water percolating through the particular soil. In soils with a high native phosphate status it is possible that the topsoils may have an "equilibrium concentration" above 100 ppb. Adsorption will then be observed from the second solution where this is the initial added concentration. The measured isotherm will then have two points below the abscissa, as shown in the example in figure 4b. This situation will be very rarely found in the lower horizons of

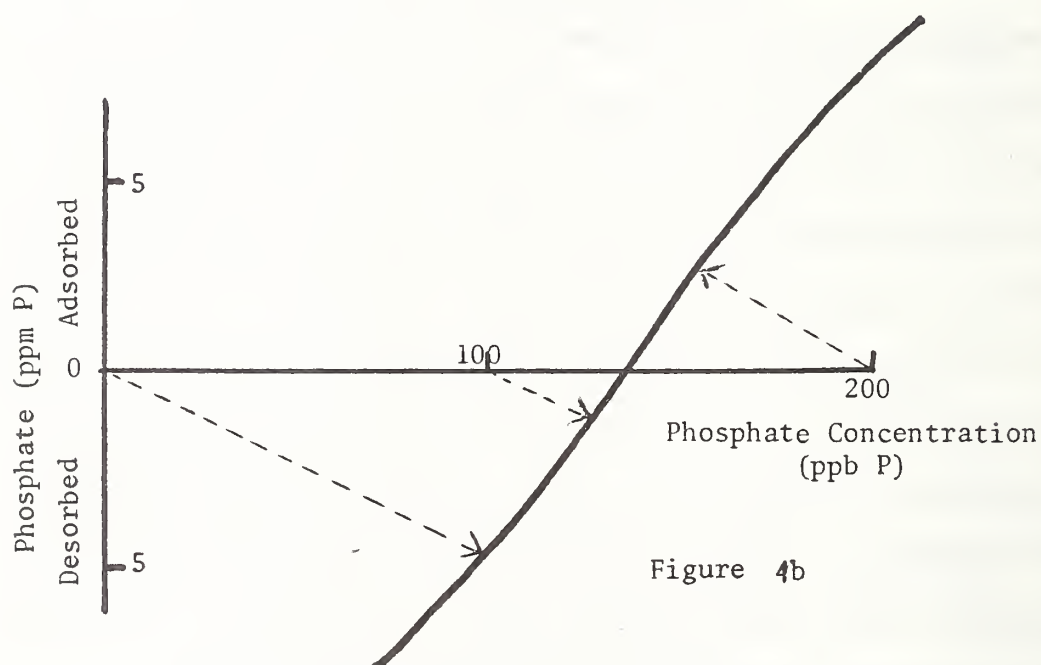
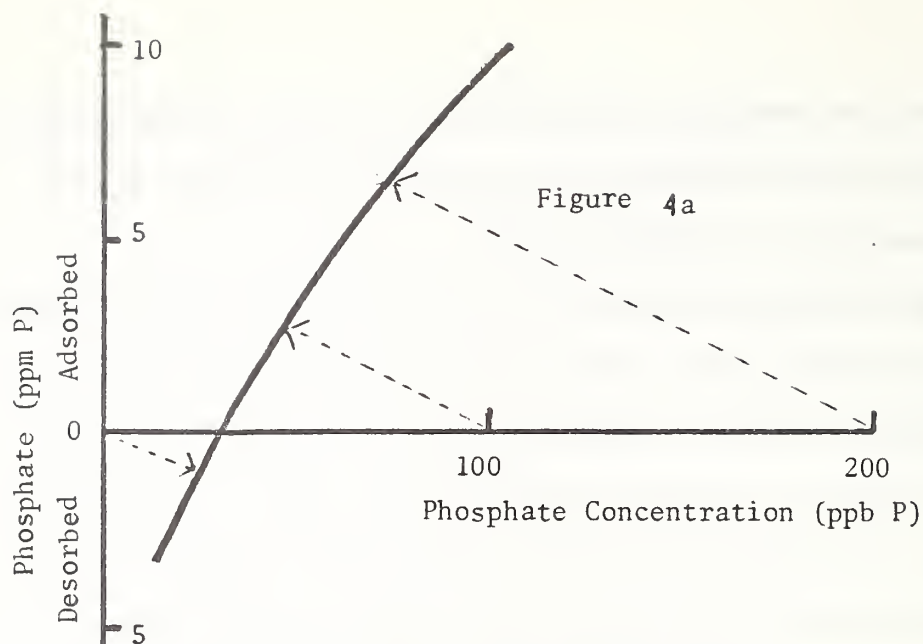


Figure 4. Isotherms for adsorption of small amounts of phosphate.

the soil. Soils which do reveal this situation should not be considered as suitable for waste-water disposal sites.

Except for the lower concentration range investigated, there is no basic difference between the adsorption isotherms determined in this way and those measured at higher concentrations, and both may be combined on the same graph if a logarithmic concentration scale is used. The slope of the isotherm may, however, be different in the different concentration ranges.

Prediction of Phosphate Adsorption in Soils

When waste water percolates downward through a soil profile adsorption will first take place in the surface layer. The phosphate concentration in the water moving out of this layer will be reduced to the value corresponding to the amount of phosphate adsorbed, as described by the adsorption isotherm.

Initially, the subsurface layers will be exposed to much lower concentrations and the subsurface adsorption will be described by the lower sections of the isotherm. As percolation continues, the concentration in the water moving through the surface layers will slowly increase, and the profile will become increasingly saturated from the top downwards. As an example, we may consider the soil whose properties are described by the isotherm B of figures 2 and 3 receiving water containing 5 ppm of phosphate. As the first water is applied the surface soil will transmit water containing less than 300 ppb of P until it has retained 145 ppm of P. For a 2.5 cm layer

of soil with a bulk density of 1.6 g/cm^3 , this is equivalent to 58kg/P hectare or 52 lbs/acre. If percolation is continued, the concentration in the water passing this layer will slowly rise to the 5 ppm level when about 225 kg/hectare or 200 lb/acre have been adsorbed. At this point the 2.5 cm layer has reached saturation equilibrium with the input water, and adsorption can only continue in the lower, still unsaturated layers.

The sequential changes that may be expected in the concentration at various depths in the soil are illustrated in figure 5. The total amount of phosphate that can be retained is clearly dependent upon the limit of concentration that can be tolerated in the water at the base of the profile. When this concentration is reached the capacity of the soil site has been saturated and it can no longer be used as a filter until remedial measures have been taken. The time that must elapse before this condition is attained depends upon a number of factors including the concentration of phosphate in the water and the rate of percolation as well as the adsorption properties of the soil. While estimates of the probable useful life of a soil can be made where these are known, the actual process should be followed by measuring the changes in the adsorption isotherms of samples from different depths as the process proceeds.

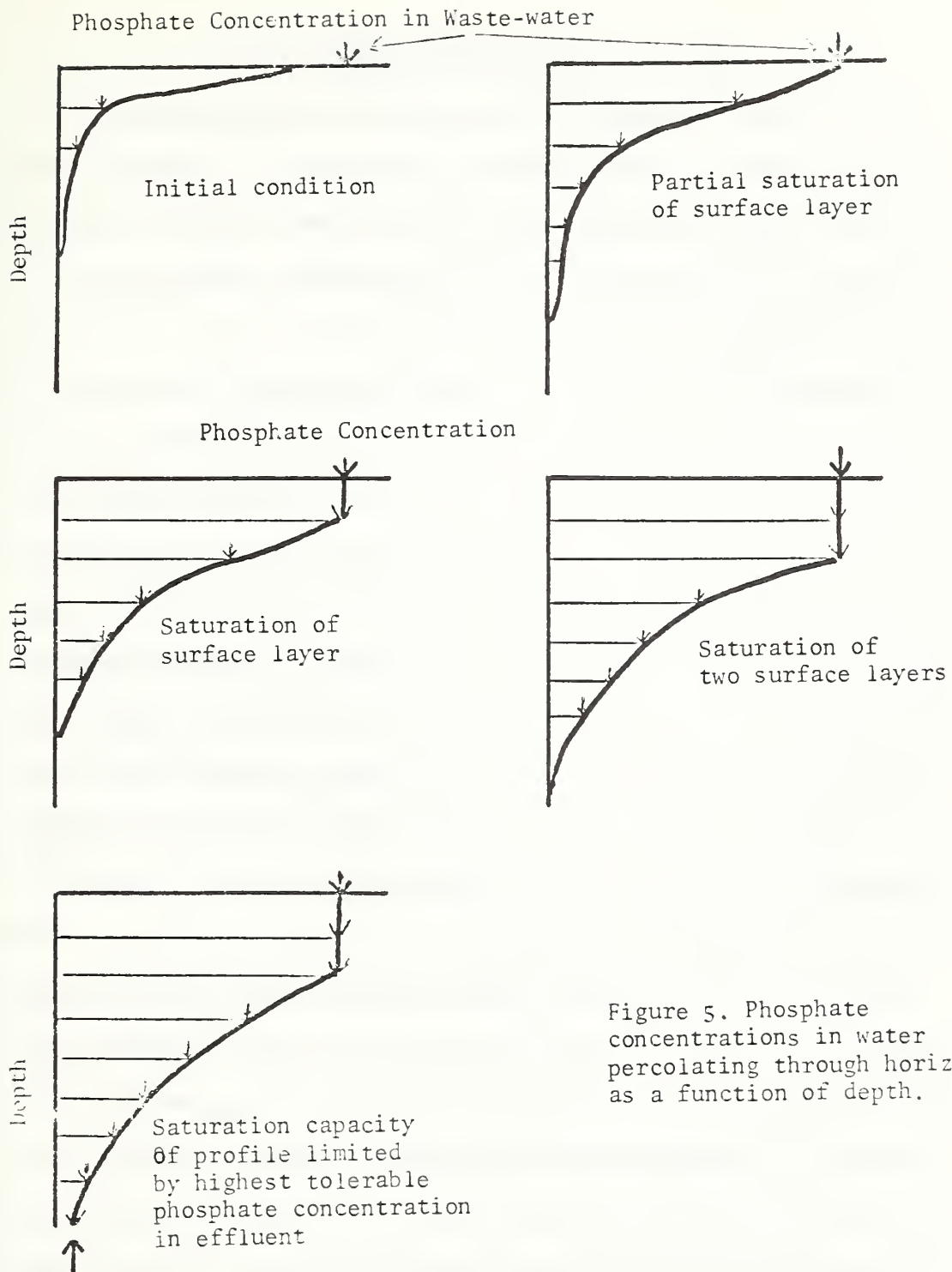


Figure 5. Phosphate concentrations in water percolating through horizons as a function of depth.

Adsorption of Organic Phosphates

The isotherms considered in the previous sections describe the adsorption of free inorganic orthophosphate ions. The presence of organically bound forms of phosphate is not however likely to invalidate the use of these except in special cases where the amounts of organic phosphates are unusually high.

The proportion of phosphorus in waste waters that is present in organic form does not usually exceed about 10 to 15% of the whole. Unless the water contains a high proportion of suspended solids this fraction will be composed of soluble material of low molecular weight. Such compounds are normally retained strongly by soil colloids. This adsorption does not interfere with the inorganic phosphate adsorption reactions which are the dominant feature of the system. Organic forms of phosphate show little tendency to move through the profile and will be retained in the upper layers. As the organic material is oxidised the phosphate present in this form is mineralized and will then be adsorbed as inorganic orthophosphate.

In special cases where the organic phosphate content of the water is known to be high its possible effects may be examined by measurements of adsorption isotherms using the actual waste water itself as the solution. Comparison of the resulting isotherms with those obtained from standard laboratory solutions of inorganic phosphate will reveal any discrepancies from the predicted isotherms. In such cases, however, special analytical techniques will be required for the determination of the amounts of organic phosphates in the initial and final solutions.

Effects of Management Practices

The maximum rate of percolation will correspond to the infiltration rate of the soil. At this rate much of the flow will take place through the larger pores so that the entire absorbing surface of the soil is not directly accessible to the flowing water. The phosphate may then move down the profile somewhat more rapidly than expected on the basis of the isotherm predictions since the rate at which the phosphate diffuses into the soil granules will become the rate limiting step. A slower rate of percolation with limited applications of water will therefore increase the effectiveness of phosphate removal.

It is also undesirable to permit the soil to remain continuously saturated with water for chemical reasons. Adsorption of phosphate upon the iron oxides depends on the presence of oxidised or ferric form of iron. Complete saturation of the soil with water results in a serious limitation of the oxygen supply. When this becomes insufficient to support the breakdown of the readily oxidised organic matter, reducing conditions may rapidly develop. The resulting production of ferrous iron will destroy the adsorbing surfaces and even release phosphate previously adsorbed upon the oxide particles. The phosphate concentrations will then rise to much higher levels than those predicted by the isotherms measured under aerobic conditions. The danger of this transition to reducing conditions is particularly great where the soil is flooded with waste water that contains soluble or suspended organic matter, since this will act as a continuous source of an active reducing agent.

The problem can be overcome by intermittent additions of wastewater so that the soil is allowed to drain and aerate in the intervals between applications. This will allow both a better distribution of the phosphate in the body of the soil and maintain the optimum chemical conditions for adsorption. Consideration should also be given to periodic surface cultivations of the soil if it shows a tendency to form a surface crust or there is a tendency for the infiltration rate to decrease as the pores become clogged.

Where it is possible to apply water by overhead spray irrigation rather than flooding, spraying procedures should always be used, since the land can often be maintained under a permanent grass cover. This management system has many advantages. If the water holding capacity of the soil is never exceeded the problems resulting from waterlogged conditions can be wholly avoided. Water consumption by the evapotranspiration of the grass reduces movement down the profile and there is an increased tendency for adsorption of phosphate in the upper layers of the soil. Uptake of phosphate by the grass from the soil will also reduce the rate at which the adsorption capacity of the soil is reached. Considerable economic advantages may also accrue if the land is used for grazing or forage production.

Maintenance of Adsorption Capacity

Since the area of land that may be available for waste water disposal may be limited it is clearly desirable to develop management practices that will extend the useful life of a site as long as

possible. The only practical means by which the adsorbed phosphate can be removed from the soil is by crop removal. It is therefore desirable that the site should be large enough that the areas to which water is applied should be rotated with others which are in crop production for one or two years. The amount of phosphate that can be removed in this way will depend upon the crop chosen and the yield attained. Values for wheat, maize, and other grains may be expected to be between 15 and 25 lbs P/acre per year. Alfalfa and leguminous crops may be somewhat higher, in the 30 lb/A range. If it is assumed that this phosphate is removed from the more labile fraction as illustrated in figure 1b, and the amount of phosphate applied annually in the water is known, an approximate balance between the gain and loss of adsorbed phosphate can be calculated. The increase in the potential life of a disposal site that can be expected by crop harvesting of the phosphate can then be readily estimated.

The introduction of resting periods between water applications will also take advantage of the slower "reversion" reactions of the adsorbed phosphate. Since the phosphate concentrations in waste waters are several orders of magnitude higher than those existing in the soil, adsorption of large amounts of phosphate from them involves a considerable disturbance of the absorption complex in the soil. Re-establishment of the equilibrium between the various levels described in figure 1b will then take place over a period of time. The introduction of resting cycles in the management of a disposal site will allow this to happen while the soil is under cultivation.

Such changes will of course be reflected in changes in the adsorption isotherms. These should therefore be redetermined on a continuing basis to monitor the effects of management practices.

Experimental Procedures

The experimental measurements necessary to evaluate the adsorption characteristics of waste disposal sites can be related to three objectives.

- (1) The monitoring of a number of possible sites to determine their suitability.
- (2) The detailed evaluation of a chosen site to develop guidelines for its management.
- (3) The monitoring of changes taking place during use.

The experimental measurements that are needed to obtain the essential information should be planned according to the scheme described in the following sections.

Measurements of the Value of Possible Sites

The basic objective is to assess the overall capacity of the soil profile as an adsorbent for phosphate. This requires a determination of the phosphate adsorption isotherms of each horizon in the profile from the surface to the water table or depth of the drain tiles through which water leaves the disposal site. The number of horizons that must be sampled for adequate characterization of the profile must be decided on the basis of the character of the soil type and its known behavior.

Experimental: A five-gram sample of each soil is added to a 100 ml volume of a solution containing 30 ppm P and 0.13% KCl (131.7 milligrams KH_2PO_4 + 1.3 g KCl per liter) and shaken for 12 - 18 hours on a reciprocating or end-over-end shaker. The suspension is then centrifuged or filtered and an aliquot of clear solution removed for analysis. The final phosphate concentration may be determined colorimetrically using the vanadomolybdophosphate yellow procedure described in the Appendix.

The results are calculated in terms of the ratio $x/(1.0 + \log c)$, where x is the weight of phosphorus adsorbed by the soil in mg/kg (ppm) and c is the final phosphate concentration in mg/liter. As described above, this ratio provides an index of the adsorption capacity in terms of the slope of the isotherm from a concentration of 0.1 ppm up to the final measured value. Together with data on the bulk density of the soil, the relative depth of the soil horizons and the anticipated composition of the waste water this ratio may be used to calculate the adsorption capacity of each soil horizon in terms of pounds/acre or kg/hectare as required.

Detailed Evaluation of Individual Sites

The procedure described above provides a single index figure of the adsorption capacity. For more detailed prediction of the behavior of an individual site adsorption isotherms should be measured over much wider concentration ranges. The principles of the experimental procedure are similar.

Experimental: Eight 5-gram samples from each soil horizon are added to 100 ml volumes of solutions containing 50, 30, 10, 3, 1, 0.3, 0.1 ppm and no phosphorus (all solutions must be 0.13% in KCl) and shaken for 12 - 18 hours. After centrifugation or filtration the final phosphate concentrations are measured colorimetrically. Since the lower concentrations may be outside the sensitivity range of the vanadomolybdophosphate method, the molybdenum blue method of Murphy and Riley should be used: this procedure is described in the Appendix.

After the final phosphate concentrations are measured, the amount of phosphate adsorbed, calculated in ppm of dry soil are plotted against $\log c$, where c is the final concentration, to give the full adsorption isotherm as illustrated in figure 3. When the final concentration in the most dilute solution is greater than the initial value, the amount of phosphorus desorbed should be plotted as a negative value below the abscissa as described above.

Measurement of isotherms for each horizon will permit a full evaluation of the expected behavior of the whole profile. Particular attention should be paid to the measurements in the lower concentration ranges in the lower horizons, since this part of the isotherm indicates the limiting concentrations that can be attained in the downward moving water.

Monitoring Management Effects

The changes in the phosphate status of the soil as a result of waste water treatment, coupled with the effects of cropping periods, can be followed by periodic re-determinations of the isotherms for each horizon. The experimental procedure should be identical with those of the previous section. Particular attention should again be paid to changes in the isotherms in the lower soil horizons.

Acknowledgment

The authors wish to acknowledge their debt to Drs. B. W. Bache and E. G. Williams of the Macaulay Institute, Aberdeen, Scotland, whose published work provided the basis for the methods recommended in this chapter.

APPENDIX

Vanadomolybdophosphate Procedure

See Jackson, M. L. Soil Chemical Analysis. Prentice-Hall, 1958. pp 151-154.

Reagents. (a) 25 g ammonium molybdate in 400 ml water: (b) 1.25 g ammonium metavanadate in 300 ml of boiling water. After cooling add 250 ml concentrated HNO_3 .

After final cooling mix solutions (a) and (b) and dilute to 1 liter.

Procedure. To 25 ml aliquot of unknown in a 50 ml volumetric flask, add 10 ml of vanadomolybdate reagent and dilute to 50 ml. After 10 minutes read color intensity with a colorimeter adjusted to 400 to 490 m μ .

A blank solution should be prepared with phosphate free solution containing 0.13% KCl. Color intensity is calibrated with a suitable set of known phosphate solutions.

Molybdenum Blue Procedure

See Watanabe, F. S. and Olson, S. R., Soil Science Society of America Proceedings, 1965, 29, pp 677-678.

Reagents. (a) Dissolve 12 g of ammonium molybdate in 250 ml of distilled water. In 100 ml of distilled water dissolve 0.291 g of antimony potassium tartrate. Add both solutions to 100 ml of 5 N H_2SO_4 (148 ml conc. H_2SO_4 per liter), mix and dilute to 2000 ml. Store in pyrex glass bottle in darkness.

(b) Dissolve 1.056 g of ascorbic acid in 200 ml of reagent (a).

This should be prepared as required and not stored for more than 24 hours.

Procedure. Dilute an aliquot of unknown containing 1 to 20 micrograms of P to about 20 ml with water in a 25 ml graduated flask and add 4 ml of reagent (b). After 10 minutes, read the intensity of the blue color with a colorimeter adjusted to 730 - 840 μm . The color intensity should be calibrated against a set of solutions of known P concentrations.

Further Reading

A Phosphate Sorption Index for Soils.

B. W. Bache and E. G. Williams. 1971.

Journal of Soil Science 22 289-301.

Wastewater Treatment and Reuse by Land Application.

U.S. Environmental Protection Agency. 1973.

Vol. I: EPA - 660/2-73 - 006a

Vol. 2: EPA - 660/2-73 - 006b.

Recommendations for Management of Potentially Toxic
Elements in Agricultural and Municipal Wastes

by

Rufus L. Chaney^{1/}

Toxic heavy metals may be present in organic waste materials which will be applied to land for fertilization or disposal purposes. We should consider as a metal-containing waste all sewage sludge, municipal refuse composts, and poultry and swine wastes from animals fed diets enriched with Cu and Zn above the small amounts needed to correct deficiencies. Processing wastes to be applied on land can also contain toxic elements from the raw materials or additions during processing.

Although every element can be toxic if it is in large enough excess over normal concentration, this discussion will be limited to the toxic elements expected to be present in municipal, animal, and most processing wastes. The trace elements of general concern are B, Cd, Co, Cr, Cu, Hg, Mo, Ni, Pb, Se, and Zn. These elements can be toxic to plants grown on waste amended soils or to the animals which consume these plants. Unusual toxic elements in industrial wastes or effluents should be known to the parties desiring land disposal thereof, and be considered as special cases if elements other than the above are involved.

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History of Metal Toxicity

Toxic amounts of Cu, Zn, and Ni have accumulated in soils from fungicides, unneeded fertilizers, sewage sludge, and industrial air and water pollution, or have occurred naturally. In most cases, the metal toxicities have occurred under intensive agricultural practices, such as orchards (Cu, Zn), vineyards (Cu), tea plantations (Ni), or vegetable fields (Cu). The toxicities can be quite expensive to alleviate. Delas (1963), and Reuther and Smith (1954) have reviewed much of the available information on metal toxicity from agricultural practices, especially Cu toxicity.

Unfortunately these experiences with inorganic forms of these trace elements can seldom be related to the land application of sludge and manure because the high organic matter and P contents of these organic wastes reduce metal injury to plants.

There has been actual field observation of metal toxicity from land disposal of effluents and sludge in England (Patterson, 1971) and other areas where toxic metals have accumulated with time to toxic levels for the crops grown. These metal toxicities to crops have generally occurred in unmanaged situations most favorable for toxicity: low pH, sandy soils, high metal sludges, and/or metal sensitive crops. The observed toxicities, because they occurred under poorly managed conditions, have generally been alleviated by liming to higher pH. However, one example of a "market garden in Somerset" involved an organic soil with very high metal content from 30 years' application of sludge at 20 tons/acre/year; in this case, apparently the only choice

was to stop growing metal-sensitive vegetable crops. This case must have been caused, in part, by use of a sludge with appreciable metal contamination, (well above the "domestic sludge" levels shown in Chapter 2).

Published reports on these earlier observations of metal toxicity generally did not supply enough information to allow interpretation and application to the metal-laden organic waste situation (little or no data on soil pH, soil C.E.C., soil organic matter content, total and plant available metals and P, and metal content of crops grown).

Potentially Phytotoxic Metals in Animal Wastes

The elements of interest are Cu and Zn. Copper at levels producing growth stimulation (antibiotic levels of 100-250 ppm Cu, much higher than needed to correct simple Cu deficiency of the ration) are being added to some swine and poultry feeds. Zn is added in higher amounts to balance the high level of Cu. The manures produced by these animals is comparable to domestic sewage sludge in levels of Cu and Zn and cannot be considered as simple organic amendments. Recently, the Food and Drug Administration has refused to permit addition of Cu as a growth stimulant (250 ppm Cu) to swine feeds because of the environmental hazards related to disposal of the manure. Up to 30 ppm Cu may be added to correct deficiency, although 6 ppm is usually adequate.

To guard against the possibility that some animals will be fed these high Cu and Zn diets, farmers using swine and poultry manures from other operators on farmland should have Cu and Zn analyses on the manure used.

Potentially Phytotoxic Elements in Sewage Sludge

The elements of concern are B, Cd, Co, Cr, Cu, Hg, Mo, Ni, Pb, Se, and Zn. Chromium is a minor problem because it occurs as Cr^{3+} in sludge and as such is not available to plants; chromate is rapidly reduced to Cr^{3+} in soils. Cadmium can be toxic to plants, but as explained below, this element is considered hazardous to the food chain if present in sludge at greater than 1.0% of the Zn content. With an acceptable 1 Cd:100 Zn ratio, added sludge Zn will be phytotoxic at added Cd levels considerably below phytotoxic. Mercury does not injure plants at the low levels added with sludge. Lead added with sludge is not phytotoxic because the sludge contains large amounts of phosphate which tie up Pb and prevent injury to and transport of Pb by plants. Cobalt seldom occurs in sludge; it is a product of specific industrial pollution. If present, it reacts much like Ni. Boron occasionally can be high, more so in composted refuse than in sludge. Boron could become phytotoxic in the same soils where we are now concerned about the B content of irrigation water. Molybdenum and selenium would not be permitted to reach phytotoxic levels in order to protect the food chain. Zinc, Cu, and Ni toxicity are readily induced by adding amounts of these metals similar to the amounts which will be added by unmanaged application of sludge. A general symptom of toxicity (at pH 5.5) is chlorosis due to Fe deficiency. Zinc, Cu, and Ni inhibit transport of Fe to the plant tops. They also inhibit root growth and absorption of macronutrients. Much information about crop injury from individual excess metals is summarized in the book

edited by Chapman (1966). Thus, the metals which commonly occur in sewage sludges and which constitute serious potential hazard to plants are Zn, Cu, and Ni.

Potentially Phytotoxic Elements in Processing Wastes

Most food processing wastes, because they arise from living plant or animal tissues with biological controls on their heavy metal composition, do not comprise any heavy metal problem. However, addition of toxic elements may occur during processing. Wastes from processing seafood such as crabs, oysters, etc., may contain high levels of metals.

Factors Controlling Phytotoxicity of Added Metals

Because the elements most likely to occur in excessive amounts when organic waste materials are added to soils are Zn, Cu, and Ni, most of the considerations presented below are based on phytotoxicity from these metals.

1. The toxic metals added:

Specific plants differ in their relative sensitivity to excess Zn, Cu, and Ni. For a single crop, these metals have different relative toxicity in different soils because of factors discussed below. Based on a number of studies, a general statement of the relative injury to plants, Cu is twice as toxic as Zn, and Ni is eight times as toxic as Zn. Chumbley (1971) introduced the Zn(equivalent) factor $[\text{ppm Zn(equivalent)} = 1 \times \text{ppm Zn} + 2 \times \text{ppm Cu} + 8 \times \text{ppm Ni}]$ to take into account the differences among metals. While the Zn(equivalent) expression of toxicity is a reasonable approximation, interactions may

occur among these metals, both greater and less than the additive Zn(equivalent).

2. The pH of the metal amended soil:

The toxic metals are much more available at lower pH values than at pH 6.5-7.0 or above. A soil metal content safe at pH 7 can easily become lethal to most plants at pH 5.5. Soil pH may actually be more important than the amount of metal added. The safest soils for metal-laden wastes would be the calcareous soils of the west. Further, land disposal of sludge or manure leads to a lowering of the soil pH due to nitrification of the high amounts of N added as ammonia in the wastes.

3. The organic matter (O.M.) content of the amended soil:

O.M. forms insoluble chelates with the toxic metals and makes them less available to injure plants. This O.M. binding is especially important for Cu and Ni. It appears that the chelation role is more important than the simple cation exchange role of the O.M. At lower pH values, the O.M. reduces metal availability relative to the same soil without the O.M. At higher pH values, O.M. addition appears to increase Zn availability (at high Zn levels). As the O.M. added with the wastes decomposes over a number of years (Mann and Barnes, 1957), its protective effect is lost. Crop rotation, green manuring, or other practices which maintain high O.M. should help reduce metal toxicity. Even additions of more sludge or manure may reduce existing metal toxicities in spite of adding more metals with the O.M. Climatic conditions affecting the decomposition of O.M. also affect the suitability of

soils for waste disposal. Thus southern soils with a generally lower O.M. content are less suitable than northern soils.

4. The phosphate content of the amended soil:

Phosphate decreases the stunting of plant growth caused by excessive levels of Zn, Cu, and Ni; P strongly counteracts metal toxicity. Phosphate also increases iron deficiency chlorosis caused by excess Cu (Spencer, 1966). Sewage sludge now contains <1-6% P and could go even higher in digested sludges from sewage treatment plants using advanced P removal processes. Manure also contains considerable P. Processing wastes, however, may be low in P. The addition of 50 tons/acre of a sludge or manure containing 4% P adds 4000 lbs. P/acre. This is an extremely large amount, and P buildup may temporarily injure some crops, e.g., soybeans. One such amendment could provide all the P needed for many years.

5. The cation exchange capacity (C.E.C.):

The C.E.C. of the soil is important in binding toxic metals. This includes both the C.E.C. of the O.M. (which strongly binds Cu and Ni by chelation), and that of the clay colloids. (The factors controlling O.M. levels in soils were discussed above.) Thus C.E.C. judgments may need to be based on the soil with a typical minimum O.M. content for soil type and climatic zone, presuming decomposition of the O.M. added in the waste. Although clay differences in C.E.C./unit weight are dramatic, and there is some indication that clays with higher C.E.C./unit weight of clay may be more effective in holding toxic metals, there appears to be no basis at this time to recommend that a soil with montmorillonitic clay is better for disposal of metal-laden wastes than a soil with illitic clays if both have the same C.E.C.

6. Reversion to lower availability:

Over time, the added metals react with the soil to become inactivitated and therefore less toxic to plants. This process has been labeled "reversion," but the process is poorly understood. Reversion has been clearly established for Zn (Brown et al., 1964), and can be quite rapid (Follett & Lindsay, 1971). The rate of reversion is lower at higher metal levels and occurs most rapidly in calcareous soils. Soil pH, and possibly PO_4 and O.M. can be related to the rate of toxic metal reversion. On a poorly managed site, the combination of rapid O.M. destruction and low pH (which slows metal reversion) may actually lead to an increase in toxic metal availability and injury. At pH 5.5-6.0, reversion of excessive levels of Zn could be a relatively slow process.

7. The plant grown on the sludge treated soil:

Plant species and even varieties differ widely in tolerance to toxic metals. Vegetable crops that are very sensitive to toxic metals include the beet family (chard, spinach, redbeet, and sugarbeet), turnip, kale, mustard, and tomatoes. Beans, cabbage, collards, and other vegetables are less sensitive. Field crops such as corn, small grains, and soybeans are moderately tolerant. Most grasses (fescue, lovegrass, Bermudagrass, orchardgrass, perennial ryegrass, etc.) are tolerant to high amounts of metals. Unusually metal-tolerant ecotypes of the grasses are found on ore outcrops containing extremely high amounts of metals (Antonovics et al., 1971).

Crops differ in their susceptibility to injury by different metals. Celery is known for its unusual tolerance to excess Cu, but it is as

sensitive to Zn and Ni as other vegetables. The 1:2:8 ratios for the relative toxicity of Zn:Cu:Ni used in the Zn(equivalent) factor are only an average of the relative tolerance for many crops, and is a good general statement. The relative tolerance of crops also varies in different soils depending upon the levels of O.M., phosphate, and pH which control the relative toxicity of Zn, Cu, and Ni. Crops also differ in the accumulation of metals in their leaves or edible parts (see below).

Food Chain Aspects of Metals Added with Wastes

Some elements constitute a hazard to the food chain if improper management of O.M. occurs. The elements of major concern are Cd and Cu, while Zn, Mo, Se, and Pb are of lesser concern. The additions to soil of very high amounts of Cr^{3+} fails to increase the Cr content of crops appreciably and comprises no hazard. The Co and Ni content of plants are increased by additions of these metals to the soils; however, the increased Co or Ni content of the crop is still so low that no hazard to the food chain exists even when the soil levels of Co or Ni are toxic to the plant. In the presence of the large amount of P added with sludge, Pb is not appreciably translocated to the tops of plants, and it is especially excluded from grain, fruits, and edible roots. Lead content of grasses tends to increase late in the growing season and over the winter; little data are available on this phenomenon in relation to the Pb added with sludge. The sludge Pb question is very different from air pollution where surface deposited Pb can enter

the food chain directly and appears to be a serious hazard. Mercury from sludge will increase soil Hg levels, but the amount of increase will be small, and the Hg is bound tightly by the sludge and soil. Further, very little Hg is trans-located to the tops of plants and soil. Mercury does not constitute the food chain accumulator in agriculture that it does in the oceans. Selenium in sludge could constitute an increased hazard where available soil Se is already high. Little data on the Se content of sludge-grown crops are available. Molybdenum from sludge and manures could lead to increased Mo levels in some crops on high pH soils. In areas where soil Mo is already high, the Mo content of the sludge should receive consideration.

Increased Zn in soils leads to substantial increase in plant Zn content. Zinc in forage is toxic to the more sensitive animals at about 1000 ppm Zn (as Zn sulfate) Because crop damage is usually economically important when the foliar Zn content reaches 500 ppm, higher forage Zn levels should be seldom encountered. Boawn and Rasmussen (1972) found that most crops suffer a 20% yield reduction at about 500 ppm Zn. However, Chaney, Baxter, and Kinlaw (unpublished) have found that the Zn content of chard leaves at a 20% yield reduction due to excess Zn is higher at low soil pH (1500 ppm) than at pH 7 as used by Boawn and Rasmussen (500 ppm). Nevertheless, because the Zn level of seed, fruit, or edible root are lower than leaves, and because yield reduction from added sludge generally will occur at lower plant Zn levels than injure animals, the food chain appears to be strongly protected.

Increased Cu in soil leads to only small increases in plant Cu content; foliar copper is not sufficiently high to be toxic to most animals until the plants show severe toxicity injury. However, sheep are unusually sensitive to Cu and could be injured by eating some forages grown on soils enriched in Cu by sludge or manure. Cu would be a special hazard to grazing animals when sludge or manure of high Cu content are sprayed on pastures (Batey et al., 1972). The leaf surface Cu is rapidly washed off with rain, but remains a hazard until this occurs. Dietary Cu and Mo interact in the ruminant. When the Mo content of forage is known to be low, expected increases in Cu content of some forages could be hazardous to sheep, while in forages of normal Mo content, Cu can be considerably higher. Liming to prevent Cu injury to plants also increases plant Mo content. Plants differ widely in accumulation of Cu; e.g., clovers accumulate substantially more Cu than grasses. Data are needed to evaluate the differences among forage crops in how rapidly sludge and manure (containing Cu, Pb, etc.) are washed off the leaves. The best advice for the interim is to apply liquid sludge and manure only to recently clipped or grazed pastures and to withhold animals until regrowth and/or rain has occurred.

Cadmium is somewhat like Zn in that increases in soil available Cd leads to increases in plant Cd. Cd accumulates in the kidney and liver over many years. Kidney damage and hypertension have been related to increased Cd levels in these organs. "Itai-Itai" disease was caused in Japan by increased dietary Cd; Cd suppressed Ca absorption and led to weak bones in older persons. The FDA expects to eventually regulate

the level of Cd in agricultural products in interstate commerce to protect against these possibilities in the United States. To help assure that sludge treated soil does not lead to a Cd hazard in the food chain, the Cd content of the sludge should be reduced as much as possible--at least to 1% of the Zn content, and if possible to <0.5% of Zn and <15 ppm Cd (see Table 1). In this way, injury to the crops from Zn excess (say at 500 ppm Zn in leaves) would occur before the Cd content of the crop constitutes a health hazard. Zn apparently competes with Cd at the sites of uptake and injury in animals, and the high amount of Zn in sludge-grown crops should serve to prevent injury. Experiments are in progress to clarify the questions about Cd and Zn relationships in crops and animals. Grain, fruit, and edible roots have a lower Zn content than the leaves of the same plants. Cd is excluded even more strongly so that the Cd/Zn ratio of grain, fruit, and edible roots are one-half to one-tenth that of the leaves. Thus, where a choice exists, these types of crops might be grown in preference to leaf crops to minimize Cd movement into the food chain. It is obvious that the concept of Cd:Zn ratio is very important in agricultural movement of Cd.

Benefits of Zn and Cu in Organic Wastes

The micronutrient Zn (and to a lesser extent Cu) is often deficient or marginally sufficient in agronomic soils. This results in low levels of Zn in crops and thus less Zn availability in the diet of animals. Fiskell et al. (1964) and Parsa (1970) have found sewage sludge to be an excellent Cu or Zn fertilizer. Miller and Lindsay (1969) found manure to be a desirable Zn fertilizer, better than an equivalent amount

of Zn sulfate. Because many people do not, or will not, supplement their low Zn diet, we should find some way to increase food Zn. Recent reports show that some teenagers' diets are low or deficient in Zn and clinical symptoms of Zn deficiency have been observed (Hambridge et al., 1972). Zn and Cu in organic wastes can be considered fertilizers when they correct such deficiencies. Sludge and manure containing Zn should increase the level of Zn in crops, and thereby help to correct the problem of low Zn in the food chain.

Recommendations for Managing Toxic Metals Added with Organic Wastes

These recommendations are based on two needs: (1) the need to limit metal additions to permit continued general agricultural use of the waste amended soil, and (2) the need to obtain sufficient agricultural benefit from sludge or manure to justify the risk from the metals contained in these wastes.

Organic wastes contain beneficial components unrelated to their toxic metal content, e.g., O.M. (of variable aerobic stability), N, P, and with manures, K fertilizer. To a lesser extent the Zn and B have fertilizer value. The toxic metals are risks which should be minimized wherever possible because of their long residence time in the surface soil. If for no other reason, the toxic metal content of sludge and manure should be minimized in order to provide the highest fertilizer benefit before the metal additions reach the maximum recommended levels.

A. Exclusion of wastes containing high levels of toxic elements:

The recommended maximum metal content of a "domestic" sludge on agricultural land shown in Table 1 (from Chapter 2) is based on

(1) the reported (see the review by Page, 1974) and our own analysis of many sewage sludges, and (2) the metal contents of sludges from sewage systems where industrial metal pollution is abated by currently available technology. Cd, Co, Cr, Ni, Pb, Hg, and B contents above these levels are clearly a result of industrial pollution. Much of the Cu and Zn comes from controllable industrial pollution, but the 2000 ppm Zn and 1000 ppm Cu in a "domestic" sludge would come principally from societal uses of these metals in pipes, tires, cosmetics, etc. The minimum levels of Zn and Cu that have been noted in digested sludge are 500 ppm Zn and 250 ppm Cu. The metal composition shown for a "domestic" sludge represents an attainable, reasonably minimum toxic-metal content for digested sewage sludge and hence a good Benefit:Risk ratio. It is therefore recommended that sludges with higher toxic-metal contents should not be applied to agricultural land. The sewage authorities involved can trace down metal polluters and enforce abatement.

Table 1. Maximum Trace Element Contents of "Domestic" Digested Sewage Sludges.

<u>Element</u>	<u>"Domestic Sludge"</u> <u>mg/kg dry weight</u>
Zn	2000
Cu	1000
Ni	200
Cd	15
Cd	1.0% of Zn
Pb	1000
Hg	10
Cr	1000

The production of sludge with low-toxic-metal-contents will result in economic savings to municipalities because they can utilize land for the ultimate disposal of their sludges. An abatement program may require several years to reach the "domestic" sludge level. While the abatement program is in progress, it may be reasonable to accept sludges with 150% of the values listed for a few years (as long as Cd content is no more than 1% of the Zn content). This would decrease the potential benefit from sludge application.

Manures should not be contaminated with toxic metals via growth stimulant levels of Cu and Zn in feeds. The FDA has refused so far to approve this addition of Cu. If a manure contains high Zn and Cu it will be no different than a sewage sludge in potential to produce a metal toxicity when applied to soil. Because the total amount of hog and poultry manure in the United States exceeds the total amount of sewage sludge, addition of growth stimulant levels of Cu and Zn to feeds actually comprises a larger long-term hazard to agricultural soils than does disposal of "domestic" sewage sludge.

B. Regulation of Application Rate:

When the toxic metal composition of the organic waste is at a minimum ("domestic" sludge levels), our next consideration is to limit toxic metal additions to levels which permit continued general farming on the amended soil. This recommendation requires some assumption or statement regarding the factors which control heavy metal toxicity. Because lowering the soil pH below 6.5 leads to extensive increases in

toxic metal availability to plants, some assumption has to be made about the pH of toxic metal-amended soils. Plant availability of toxic metals is sharply reduced in soils limed to a pH of 6.5 with a considerably smaller reduction of availability above this pH. Thus, it is necessary to presume that toxic metal amended soil used for food crops will be maintained at pH 6.5 or above at all times.

Chumbley (1971), basing his comments on a considerable body of British experience with metal toxicity, recommended that no greater than 500 lbs Zn(equivalent) of toxic metals/acre be added to soils (at pH >6.5) in any 30-year period. His recommendation did not take into account any of the important soil variables except pH. Leeper (1972) summarized published data showing that one could add toxic metals up to about 5% of the C.E.C. before phytotoxicity occurred (at pH >6.5). [In the case of a soil of C.E.C. = 15, 250 ppm Zn equals about 5% of the C.E.C.] Our research also suggests that the 5% of C.E.C. of unamended soil figure (as metal sulfates) could still allow appreciable injury to toxic metal sensitive crops at pH 5.5; at pH >6.5, the Zn and Cu would not be phytotoxic. After several years, however, reversion could have occurred so that the soil pH could drop to 6 and not cause yield reduction of sensitive crops because of reversion.

Toxic metal additions to agricultural soils should not exceed Zn(equivalent) levels equal to 5% of the C.E.C. of the unamended soil (at pH >6.5). When our knowledge of the reversion process for toxic metals increases, this limit will likely be set higher. At the present state of knowledge, 5% of C.E.C. appears to be sufficiently protective

of our agricultural soil resources. To convert this to recommended maximum tons/acre for any particular sludge, the following Equation 1 may be used:

$$(1) \text{ Total sludge (dry tons/acre) } = \frac{16300 \times \text{C.E.C.}}{\text{ppm Zn} + 2(\text{ppm Cu}) + 8(\text{ppm Ni}) - 300}$$

C.E.C. = cation exchange capacity of the unamended soil in meq/100 g as determined by sum of cations or equivalent method

ppm = mg/kg dry weight of sludge

In this equation, the Cu and Ni contents are converted to Zinc (equivalent) which takes into account the greater phytotoxicity of Cu and Ni; other heavy metals are not considered important in this plant toxicity index. The -300 adjusts for the addition to the soil of some exchange capacity in the inorganic matter of the sludge or manure. This recommendation applies only to soils which can be adjusted and held at a pH of 6.5 or greater for a period of at least 2 years after sludge or manure application. Otherwise, lower total applications of metals would be recommended.

The soil C.E.C. is the best ordinarily measured character of soils which relates to the ability of a soil to hold excess metal or protect plants. It is an indirect measure of O.M. and oxides. As research clarifies these factors in reversion, perhaps a factor which directly controls available metals can be used in these calculations.

Extractable values determined by the DTPA-ETA method, the 0.1 N HCl method or the 0.050 N HCl + 0.025 N H₂SO₄ method may be used;

multiply extractable Cu X 2 and extractable Ni X 8. Sites where sludge is to be applied should have soil analysis to determine whether metals had been added previously (e.g., from manure, sludge, pesticides, etc.).

If it is necessary to take previous metal additions into account, Equation 2 may be used:

(2) Total sludge (dry tons/acre) =

$$\frac{\text{C.E.C.} \times 32.7 \times 10^4 \left[0.05 - \frac{\text{extractable ppm Zn (equivalent)}}{327 \times \text{C.E.C.}} \right]}{\text{ppm Zn} + 2(\text{ppm Cu}) + 8(\text{ppm Ni}) - 300}$$

C. General Comments:

Metal leaching should be a problem only on acid, low C.E.C. soils with strongly acid, and sandy subsoils.

Because metal toxicity generally injures roots more than tops, in drouthy areas the first symptom of metal toxicity may be wilting.

In soils where rooting depth is limited to the metal amended area, relatively more injury would occur if the pH dropped to 5.5 than in soils with no limitation of rooting depth. Foliar chlorosis is the symptom of metal excess only when the plant roots are unable to reach below the metal toxic area to obtain available Fe (based on Florida experience with excess Cu).

If dried or dewatered sludges are poorly mixed into the soil during the first season, plant roots may avoid the toxic clumps of sludge. Metals are strongly bound by the O.M. in the clump; thus, less injury will occur than when the sludges are uniformly mixed into the soil. The potential for metal injury would increase with further mixing.

Where the soil is ordinarily tilled to greater than 6 inches, higher sludge applications may be appropriate. In these cases, the pH of the sludge-soil mixture below 6 inches should be above pH 6.5 for 2 years after application. This pH maintenance recommendation applies especially where sludge will be placed below ordinary tillage depth (e.g., with trenching).

Reversion of toxic metals at high rates of metal application appears to be strongly related to pH although at low levels of metals the pH effects are less important. Maintaining a soil at \geq pH 7 for 3 years after the sludge amendment could lead to extensive reversion. If the soil pH falls after this time, plant injury is the result of the agricultural management practices used (e.g., use of $(\text{NH}_4)_2\text{SO}_4$ fertilizers without lime additions). Lowering of pH \leq 5.5 for several years, or prolonged submergence could lead to reversal of the reversion produced earlier by proper pH management.

One-shot additions of sludge to the maximum recommended metals addition should require initial pH management to \geq 7 in order that the pH at equilibrium be \geq 6.5.

Toxic metal availability and reversion can be measured with the DTPA-TEA (pH 7.3) extraction method of Lindsay and Norvell (see Follett and Lindsay, 1971). The 0.1 N HCl method underestimates Cu and other metals bound strongly by soil O.M. Insufficient baseline data for DTPA-extractable metals vs. pH, etc., are available at this time to determine what is the plant toxicity level for the specific

metals, but this data should be available relatively soon for Zn, Cu, Ni, and Cd.

Toxic metals added with sewage always initially appear to be less toxic than predicted from studies where soluble metal salts were added. The presence of high levels of phosphate counteracts the expected injury from Cu and Zn. The O.M. added as a part of the sludge temporarily increases the ability of the soil to hold metals in unavailable forms. The sludge contains 1-5% Fe, much more than acid, sandy soils. The localized distribution of sludge in the soils allows plant roots to avoid metal toxic areas. Much of the sludge metals can be present as insoluble sulfides and carbonates; these become more available with time, even at pH 6.5 (sulfides are oxidized to soluble sulfates). Recommendations for permissible levels of toxic metals must be based on the long-term rather than these initial observations.

When soils must be limed to reach pH 7 as part of a sludge disposal program, care should be taken to avoid Ca/Mg imbalance based on analysis of the sludge and soil.

In many sludge utilization schemes, the sludge may be applied annually to supply the N or P fertilizer requirement for the crop grown. Thus, the metals will build up to the maximum recommended levels over a period of years---perhaps as long as 40-50 years with a "domestic" sludge. Monitoring of DTPA-extractable toxic metals should provide the necessary tool to follow accumulation and reversion of the metals in the soil.

Sludge treatment by the sewage treatment plant may effect the need for management on the farm. Lime is often added to facilitate vacuum filtration and leads to a calcareous sludge. Chlorination of the sludge generates very acidic sludge which will require liming in nearly every agricultural use. Some heat drying processes appear to partially inactivate the sludge metals.

On some sites, municipalities will desire to dispose of maximum rates of sludge (or sludges of higher metal content than shown in Table 1) for as long as possible. In this case, one can select management conditions which would allow maximum agricultural production and protection of the food chain, presuming that appropriate management and monitoring practices are used. Foliar and soil analysis would show when the crops should be changed to crops of greater metal tolerance or to protect the food chain. Ground cover can always be maintained, even if special strains of grasses have to be used. Soils of initially marginal value may be found (e.g., gravel or coal strip mine spoil) which can be leveled and then far more sludge applied than is safe for general agriculture. Even here, sludges with Cd >1% of the Zn content should not be used. Grain and grass crops may be used with good pH management to obtain productive agricultural use of the site. Grazing animals often ingest soil and the ingested soil from such a site may provide them with too much Cu, Pb, or Cd. This high-rate sludge disposal practice should be limited wherever possible because the nutrients are being squandered on a limited area. These sites may require monitoring of soils, crops, and groundwater; they probably should remain continuously

under control of the responsible municipality. Perhaps this practice should be considered in the same category of use as land fills.

REFERENCES

1. Allaway, W. H. 1968. Agronomic controls over the environmental cycling of trace elements. *Adv. Agron.* 20:235-274.
2. Andersson, A. and K. O. Nilsson. 1972. Enrichment of trace elements from sewage sludge fertilizer in soils and plants. *Ambio* 1:176-179.
3. Antonovics, J., A. D. Bradshaw, and R. G. Turner. 1971. Heavy metal tolerance in plants. *Adv. Ecol. Res.* 7:1-85.
4. Batey, T., C. Berryman, and C. Line. 1972. The disposal of copper-enriched pig-manure slurry on grassland. *J. Br. Grassld. Soc.* 27: 139-143.
5. Boawn, L. C. and P. E. Rasmussen. 1971. Crop response to excessive zinc fertilization of alkaline soil. *Agron. J.* 63:874-876.
6. Brown, A. L., B. A. Krantz, and P. E. Martin. 1964. The residual effect of zinc applied to soils. *Soil Sci. Soc. Amer. Proc.* 28: 236-238.
7. Chaney, R. L. 1973. Crop and food-chain effects of toxic elements in sludges and effluents. *In Proc. Jt. Conf. Recycling Municipal Sludges and Effluents on Land. Nat. Assoc. St. Univ. and Land Grant Coll., Wash., D. C., p 129-141.*
8. Chapman, H. D. Diagnostic Criteria for Plants and Soils. Univ. Calif., Div. Agr. Sci. 1966.
9. Chumbley, C. G. 1971. Permissible levels of toxic metals in sewage used on agricultural land. *A.D.A.S. Advisory Paper No. 10, 12 pp.*
10. Delas, J. 1963. The toxicity of copper accumulating in soils. *Agrochimica* 7:258-288 (in French).
11. Fiskell, J. G. A., P. H. Everett, and S. J. Locascio. 1964. Minor element release from organo-N fertilizer materials in laboratory and field studies. *J. Ag. Food Chem.* 12:363-367.
12. Follett, R. H. and W. L. Lindsay. 1971. Changes in DTPA-extractable zinc, iron, manganese, and copper in soils following fertilization. *Soil Sci. Soc. Amer. Proc.* 35:600-602.
13. Fulkerson, W. and H. E. Goeller (eds). 1973. Cadmium. *The dissipated element.* ORNL-NSF-EP-21. 473 pp.

14. Hambridge, K. M., C. Hambridge, M. Jacobs, and J. D. Baum. 1972. Low levels of zinc in hair, anorexia, poor growth, and hypogensia in children. *Pediat. Res.* 6:868-874.
15. Hinesly, T. D., R. L. Jones, and E. L. Ziegler. 1972. Effects on corn by applications of heated anaerobically digested sludge. *Compost. Sci.* 13:26-30.
16. King, L. D. and H. D. Morris. 1972a. Land disposal of liquid sewage sludge: I. The effect on yield, in vivo digestibility, and chemical composition of coastal bermudagrass (Cynodon dactylon L. Pers). *J. Environ. Qual.* 1:325-329.
17. King, L. D. and H. D. Morris. 1972b. Land disposal of liquid sewage sludge: II. The effect on soil pH, manganese, zinc, and growth and chemical composition of rye (Secale cereale L.). *J. Environ. Qual.* 1:425-429.
18. Leeper, G. W. 1972. Reactions of heavy metals with soils with special regard to their application in sewage wasges. Department of the Army, Corps of Engineers under Contract No. DACW73-73-C-0026. 70 pp.
19. Mann, H. H. and T. W. Barnes. 1957. The permanence of organic matter added to soil. *J. Agr. Sci.* 48:160-163.
20. Miller, B. F., W. L. Lindsay, and A. A. Parsa. 1969. Use of poultry manure for correction of Zn and Fe deficiencies in plants. *Proc. Conf. Animal Waste Management*, Cornell Univ.
21. Page, A. L. 1974. Fate and effects of trace elements in sewage sludge when applied to agricultural lands. A literature review study. U.S. Environ. Protection Agency Rpt. No. EPA-670/2-74-005. 108 pp.
22. Patterson, J. B. E. 1971. Metal toxicities arising from industry. In Trace elements in soils and crops. *Min. Agr. Fish. Food Tech. Bull.* 21:193-207.
23. Parsa, A. A. 1970. Solid wastes as Zn fertilizers. Ph.D. Thesis. Colo. St. Univ. 47 pp.
24. Reuther, W. and P. F. Smith. 1954. Toxic effects of accumulated copper in Florida soils. *Soil Sci. Soc. Fla. Proc.* 14:17-23.
25. Spencer, W. F. 1966. Effect of copper on yield and uptake of phosphorus and iron by citrus seedlings grown at various phosphorus levels. *Soil Sci.* 102:296-299.
26. Webber, J. 1972. Effects of toxic metals in sewage on crops. *Water Pollut. Control* 71(4):404-413.

ORGANIC MATTER DECOMPOSITION AND OXYGEN RELATIONSHIPS

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INTRODUCTION

Concepts and considerations for land spreading of organic wastes have changed dramatically in recent years. Where animal wastes have been traditionally applied at rates ranging from 5 to 10 tons per acre, these and other materials, including sewage sludges and processing wastes, are now often applied at rates 10 to 50 times higher. The actual rate of application would depend on the specific objectives involved; i.e., (i) whether to utilize the wastes for their plant nutrients in crop production or (ii) to use the land only as a disposal site. Rapid decomposition of organic wastes and mineralization of their organic nitrogen by the soil microflora depend on an adequate supply of molecular oxygen. However, the high biochemical oxygen demand (BOD^{2/}) of many wastes, high loading rates, and the mode of application often create anaerobic environments which lead to significant changes in microbial metabolism. The chemical nature of the end products of microbial metabolism under anaerobiosis is quite different than that found in a highly aerobic or oxidized system. Excessive loading rates can result in a period of extended anaerobiosis and the production of toxic metabolites in sufficient concentration to inhibit seed germination and retard plant growth for some time, even after aerobic conditions are resumed.

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^{2/} Biochemical oxygen demand is the oxygen consumed by microorganisms in the process of decomposing organic wastes under standard conditions and during a specific period of incubation. It is expressed in terms of mg O₂/liter or ppm and is an indication of the readily oxidizable materials present.

FACTORS AFFECTING THE RATE AND EXTENT OF DECOMPOSITION OF CROP RESIDUES AND ORGANIC WASTES IN SOIL

A number of factors can affect the decomposition of various organic substrates when applied to soil. These can be grouped into substrate factors and soil factors. Substrate factors would include (a) chemical composition, (b) C:N ratio, (c) lignin content, (d) particle size or state of subdivision, (e) nature of the indigenous microflora, and (f) BOD. Soil factors would include (a) temperature, (b) oxygen supply, (c) moisture, (d) pH, (e) available nutrients (N, P, K, S), and (f) soil texture and structure. The rate of decomposition of a waste or residue will depend primarily on its chemical composition, and on the chemical and physical properties of the soil which determine the nature of the environment for microbial growth and metabolism. The exact physico-chemical nature of this environment will determine the specific types and numbers of soil microorganisms available to decompose the substrate. It should be recognized that those factors which have the greatest effect on microbial growth and metabolism will have the greatest potential for altering the rate and extent of substrate decomposition in soil.

A brief discussion of just how some of these factors can affect residue or substrate decomposition follows:

a. The C:N Ratio -- Because crop residues contain the same (about) amount of carbon (40% on a dry weight basis), their N contents are often compared on the basis of C:N ratios.

Thus, a low N content or wide C:N ratio is associated with slow decomposition. Although the N content or C:N ratio of crop residues

or other organic wastes can be useful in predicting decomposition rates, they should be used with some caution since the C:N ratio says nothing about the microbial availability of the carbon or nitrogen.

b. Lignin Content -- The rate of decomposition of crop residues and some animal wastes is often proportional to their lignin content. Wastes having a high lignin content decompose more slowly than those containing a small amount of lignin. Some researchers have suggested that the lignin content of some wastes may be a more reliable parameter for predicting the rate of decomposition than the C:N ratio.

c. Temperature -- Changes in temperature can alter the species composition of the soil microflora.. Different organisms have different temperature optima for maximum growth and activity. Maximum decomposition rates for residues and wastes in soil are generally obtained in the range of 30⁰ to 35⁰ C.

d. O₂ Supply -- Maximum decomposition rates are dependent upon an adequate supply of molecular O₂. While many soil bacteria can grow under anaerobic conditions, though less actively, most fungi and actinomycetes do not grow at all. Thus, residues decompose more slowly and are subject to incomplete oxidation under conditions of soil anaerobiosis.

e. Soil Moisture Content -- Soil microorganisms are affected differently depending on the moisture regime, which may greatly influence the rate and extent of residue decomposition. For example, maximum bacterial growth and activity occurs in soils at high water potentials (wet soils), but is noticeably decreased at about -3 bars (drier soils) and markedly so at -15 bars (very dry soils; i.e., the wilting point).

However, fungi tend to grow and survive in soils at much lower water potentials (dry soils) where bacteria are less active. In wet soils, where bacteria tend to flourish, fungal growth is often suppressed because of the greater competitive and antagonistic advantage of bacteria under these conditions.

f. Soil pH -- While different types of soil microorganisms have different pH optima for maximum growth, the optimum pH range for rapid decomposition of wastes and residues is 6.5 to 8.5. Bacteria and actinomycetes have pH optima near neutrality and do not compete effectively for nutrients under acidic conditions, which would explain why soil fungi often become dominant at lower pH values.

CHEMICAL COMPOSITION AND NITROGEN RELATIONSHIPS

Most of the waste and residue materials applied to soil consist of a water-soluble fraction and a water-insoluble fraction with vastly different rates of decomposition. The water-soluble fraction is comprised of sugars, starches, organic acids, pectins, tannins, and some protein, which are subject to early, rapid utilization as carbon and energy sources by the soil microflora. At a less rapid rate, compounds in the water-insoluble fraction consisting of hemicellulose, cellulose, fats, waxes, resins, and oils would be utilized. Lignin, the constituent least susceptible to degradation would tend to persist and accumulate in the soil organic matter or humus.

During decomposition of wastes and residues low in N, the C:N ratio tends to decrease with time. For example, the C:N ratio of undecomposed

corn stalks is about 50:1, but after 6 to 8 months this residue would approach a C:N ratio similar to that reported for most native soil organic matter; i.e., about 10:1 or 12:1. The decrease in the C:N ratio results from the net loss of carbon as CO_2 during decomposition, while most of the N (including available inorganic soil N) is quickly assimilated and immobilized in microbial cellular material until the C:N ratio is sufficiently narrowed. Thus, the percentage of N in residual plant material steadily increases as decomposition proceeds.

The total N content of some characteristic organic wastes and residues is shown in Table 1. Those materials which contain more than 1.5% N would ordinarily need no supplemental fertilizer N or soil N to meet the demands of soil microorganisms during decomposition. Although there is some variation, depending on the chemical composition of a particular waste and the rate of decomposition, the 1.5% is a guideline which is generally accepted. Since the N content of most mature crop residues, some cattle feedlot manures, municipal waste composts, some paper mill sludges, and most cannery wastes are considerably less than 1.5%, there is immediate concern that their application to soil could induce an N deficit during decomposition, leading to microbiological immobilization of inorganic soil N for extended periods. The addition of supplemental inorganic fertilizer N in such cases will actually accelerate the rate of residue decomposition. Ultimately, however, after 6 months or a year, there would be little difference in the extent of decomposition whether N is added or not. The total residual carbon which enters the humus fraction would be about the same, with or without N. Thus, supplemental inorganic N can be added to some

wastes and residues of low N content to accelerate their decomposition or, in the case of some crop residues, to ensure against any deficiency of soil N if another crop is soon to be planted.

On the other hand, wastes such as poultry manure and sewage sludges contain much higher levels of N and would be expected to release inorganic N, probably as ammonia, soon after application to soil. The single most important consideration which may limit the loading rates for land spreading of these wastes is their high N content. Excessive loading rates (>50 tons of dry waste/acre) of these wastes on land could soon result in the pollution of surface and ground waters by nitrate-N, through runoff and leaching, respectively.

THE INDIGENOUS MICROFLORA OF WASTES AND BOD RELATIONSHIPS

It is often overlooked that most organic wastes and residues inherently contain populations of indigenous microorganisms including bacteria, fungi, actinomycetes, and protozoa. For example, in fresh animal manures microbial cells and cells from the intestinal lining of the animal make up about 40% of the feces. A fresh cow manure slurry may contain 10^6 anaerobic bacteria, 10^5 coliform bacteria, 10^6 enterococci bacteria, and 10^5 fungi per ml of suspension. Coliform bacterial counts as high as 18 billion excreted per animal per day are not unusual. This, along with the high percentage of digestible materials involved, explains why animal manures are characterized by such high BOD values, which may run as high as 100,000 mg O_2 /liter (i.e., ppm), respectively, in a fresh cow manure slurry of feces and urine. Animal wastes carried

in runoff from feedlots are considerably lower in BOD, ranging from 100 to 10,000 mg O₂/liter, depending on the extent of dilution and degree of deterioration of the waste. Decomposition of animal manures during extended periods of storage results in the loss of carbon as CO₂, concentration of inorganic nutrients, loss of N as NH₃, and denitrification of nitrate and nitrite. This would explain why the N content of cow manure from some cattle feedlots is little more than 1% (Table 1).

Other wastes such as sewage sludge, cannery wastes, and green or mature crop residues are also known to have rather high indigenous populations of microorganisms. The exact role of these organisms in the decomposition of wastes and residues following application to soil is not known. Hopefully, this will be the subject of some future research investigations.

NUTRITIONAL REQUIREMENTS AND TYPES OF MICROBIAL METABOLISM WHICH INFLUENCE OXYGEN RELATIONSHIPS

The rate and extent of organic waste decomposition and the rapid reduction of BOD depend on a continuous supply of available oxygen. Diffusion of molecular O₂ into soils subjected to excessive loading rates is often impeded because of excess moisture associated with the waste or physical restrictions from heavy surface applications. Thus, conditions which exist in waterlogged soils are often similar to those where upland soils are amended with heavy loadings of high BOD wastes.

Table 1--Total nitrogen content of some organic wastes and residues

	N-percent of dry material
Cattle feedlot manure-----	1.2 - 2.0
Fresh cow manure, litter, free-----	2.4
Poultry manure-----	3.5 - 5.0
Municipal waste compost-----	1.0
Paper mill sludges-----	0.2 - 2.3
Sewage sludges -----	2.0 - 6.0
Cannery wastes -----	0.5 - 1.8
Corn stover -----	0.9
Wheat straw-----	0.5

The development of more efficient and effective methods for spreading of organic wastes and residues on land depend on a clear understanding of the nutritional requirements and types of microbial metabolism which influence O_2 relationships, a brief discussion of which follows:

Aerobic respiration by soil microorganisms involves oxidation-reduction reactions in which molecular O_2 serves as the ultimate electron acceptor, while an organic substrate (i.e., the waste or residue) functions as the electron donor or energy source. If O_2 is available for active decomposition, the system will be dominated by an array of different soil microorganisms, including bacteria, antinomycetes, and fungi.

Anaerobic respiration by soil microorganisms includes biological oxidation-reduction reactions in which inorganic compounds, rather than molecular O_2 , serve as the ultimate electron acceptor. Here again, the organic waste or residue serves as the electron donor or energy source. Thus, if O_2 is depleted during active decomposition, as is often the case, the system will be dominated by facultative anaerobic bacteria. Most soil bacteria are in this category; that is, they can readily utilize molecular O_2 as an electron acceptor in aerobic respiration, but under anaerobiosis they are capable of utilizing nitrate (NO_3^-), manganic (Mn^{4+}), and ferric (Fe^{3+}) ions as electron acceptors, thereby reducing them to nitrite (NO_2^-), manganous (Mn^{2+}), and ferrous (Fe^{2+}), respectively. Under strict anaerobic conditions, obligate anaerobic bacteria of the genus *Desulfovibrio* utilize sulfate (SO_4^{2-}) as an electron acceptor, reducing it to sulfide (S^{2-}). These organisms are capable of utilizing organic acids as electron donors in this reduction. Most soil antinomycetes and fungi are obligate aerobes and would not be active under soil anaerobiosis.

Fermentation includes energy-yielding reactions performed by select groups of obligate and facultative anaerobic bacteria in which organic compounds serve as both electron donors and electron acceptors. Fermentation occurs mainly in anaerobic, substrate-amended soils, particularly under extremely reduced conditions. These reactions result in the accumulation of incompletely oxidized organic compounds, including an array of organic acids and alcohols. A special example of fermentation is performed by the *methane bacteria* which are obligate anaerobes capable of degrading organic acids to yield methane. This type of reaction might be expected to occur in soils amended with excessive loadings of high BOD wastes or in sanitary landfills.

Succession of Reductive Processes in Soils Amended

With Organic Wastes and Residues

There is a great similarity in the events which occur in poorly drained or waterlogged soils compared with soils receiving heavy loadings of high BOD wastes. As the level of molecular O_2 steadily decreases during waste decomposition, the type of microbial metabolism changes successively according to the oxidation-reduction state (i.e., redox potential of Eh^3) involved, ranging from aerobic respiration in the presence of molecular O_2 to methane fermentation under complete anaerobiosis. The succession of types of microbial metabolism occurring in a soil after waterlogging relative to the soil Eh (Table 2) is the same that is likely to occur when wastes are applied to soils at excessive loading rates.

³ Eh is an expression of the electron density of a system. As a system becomes increasingly reduced there would be a corresponding increase in the electron density, resulting in a progressively increased negative potential.

Table 2--Succession of events related to the redox potential which may occur in a waterlogged soil or a soil receiving excessive loading rates of organic wastes.

Period of incubation	Stage of reduction	System	Redox potential (millivolts)	Nature of microbial metabolism	Formation of organic acids
Early	First stage	Disappearance of O_2	+600 to +400	Aerobes	None
		Disappearance of NO_3^-	+500 to +300	Facultative anaerobes	Some accumulation after addition of organic matter
		Formation of Mn^{2+}	+400 to +200		
		Formation of Fe^{2+}	+300 to +100		
Later	Second stage	Formation of S^{2-}	0 to -150		Rapid accumulation
		Formation of H_2	-150 to -220	Obligate anaerobes	Rapid decrease
		Formation of CH_4	-150 to -220		

Two stages are evident in the transition of microbial metabolism governing the reductive processes. During the first stage, which occurs early in the incubation period, oxidative decomposition of wastes and residues proceeds through the activity of aerobic and facultative anaerobic microorganisms. Carbon dioxide and NH_3 are evolved rapidly with little or no accumulation of incompletely oxidized organic compounds. Soon after waterlogging, or the application of excessive loadings of high BOD wastes, molecular O_2 disappears from the system and the Eh begins to decline. In fairly rapid succession nitrates disappear, mainly as a result of biological denitrification, and the formation of manganous (Mn^{2+}) and ferrous (Fe^{2+}) ions follows. During the first stage, Eh drops from +600 to about +100 millivolts (mV).

The second stage in the reduction process may occur somewhat later in the incubation period and is characterized by the reduction of sulfate (SO_4^{2-}) to sulfide (S^{2-}), which can result in the formation of H_2S . Sulfide formation is followed by the appearance of products of incomplete organic matter decomposition; i.e., organic acids, molecular hydrogen, and methane. Organic acids accumulated at this time soon disappear due to their utilization as electron donors and as a source of carbon by sulfate-reducing and methane-producing bacteria. The amount of molecular hydrogen produced varies greatly while the amount of sulfide and methane continues to increase as the Eh drops to -200 mV.

THE POSSIBLE CONSEQUENCES OF EXTENDED SOIL ANAEROBIOSIS FROM LAND SPREADING OF WASTES

The maximum amount of a particular waste or residue which can be applied to a soil without serious environmental consequences is not known. We do have, however, considerable information which suggests that when wastes are

applied to land at more reasonable or acceptable loading rates (10 to 30 tons/acre of dry waste), as compared with excessive loading rates (>50 tons/acre), and providing the soils are well drained but not sandy, there have been few problems concerning nitrate runoff and leaching or subsequent impairment of the soil for crop production. Where researchers have attempted to establish maximum loading capacities for soils using loading rates often far in excess of 50 tons/acre, there is evidence of associated environmental problems, particularly that of excess nitrogen, although heavy metals are also of some concern in the case of sewage sludges.

The data in Table 2 provide a good indication of what we can expect when a soil is loaded beyond its capacity for sustained aerobic decomposition. Possibly a more helpful understanding of the consequences which can develop from excessive loadings of a high BOD waste compared with lower, acceptable loadings is shown in Figure 1. With acceptable loadings, O_2 balance is maintained as rapid aerobic decomposition is ensured. Thus, the end products of decomposition are inorganic carbon, nitrogen, and sulfur compounds. However, with excessive loadings of wastes, rapid O_2 depletion occurs, and there is an equally rapid transition to a highly reduced anaerobic state where the rate and extent of decomposition is slowed considerably, and where undesirable chemical changes occur resulting in the production of obnoxiously odorous compounds (amines, mercaptans, and H_2S) and compounds having pronounced phytotoxic effects (NH_3 and H_2S). Depending on the extent of overloading, the soil may remain in an anaerobic condition for an extended period.

The relationship between loading rate and application frequency relative to the probably period of soil anaerobiosis is shown in Figure 2, which

illustrates the effects of frequent, successive applications of wastes at acceptable loading rates, compared with single slug applications at an excessive loading rate, on changes in the soil redox potential. Excessive loading rates of high BOD wastes often lead to a rapid depletion of O_2 and equally rapid decrease in soil Eh to an undesirably reduced state where it remains for a matter of months, after which a slow upward trend may occur. On the other hand, a proper balance of loading rate and application frequency can maintain the soil Eh within a more desirable range, where extensive reduction does not occur and where the likelihood of odorous and phytotoxic end products arising is minimal. While some downward deflection of Eh can be expected even with acceptable loading rates, the system will tend to recover rapidly because it has not been loaded excessively or beyond its limits. Thus, the cyclical Eh pattern allows successive and more frequent applications as the Eh moves back into the aerobic range.

In summary, most soils have a tremendous capacity to decompose organic wastes. The soil provides (i) a medium for dilution of waste concentrates, (ii) a highly important buffering system, and (iii) a potentially active microflora for rapid and sustained decomposition. Where wastes are applied at acceptable loading rates (10 to 30 tons/acre, dry weight basis), the problems associated with waste decomposition including extended anaerobiosis and undesirable end products are indeed minimal. However, soils do have some limits as to how much waste they can accommodate at any one time, and if loaded excessively (>50 tons/acre) there will be certain deleterious effects of rapid O_2 depletion, extended anaerobiosis extensive reduction, and the accumulation of odorous and phytotoxic end products which could impair the soil for crop production for some time.

SPECIFIC GUIDELINES

1. The maximum amount of a waste or residue that a soil will accommodate for sustained aerobic decomposition without serious environmental hazard or impairment of the soil for crop production is not known.

2. There is considerable information which suggests that when wastes are applied to land at acceptable loading rates (10 to 30 tons/acre, dry weight basis), and providing the soils are well drained and not sandy, there are few problems concerning nitrate runoff and leaching.

3. Where researchers have attempted to establish maximum loading rates for the disposal of wastes (rather than utilization as a source of nutrients), problems of excess inorganic nitrogen have often occurred.

4. The single most important consideration in the land spreading of high BOD wastes is the N content. Approximately 50% of the total N (mainly organic N) contained in the waste will be mineralized during the first year; 25% the second year; and 12% the third year. Thus, environmental problems related to nitrate runoff and leaching are predictable based on the N content.

5. Excessive loading rates (>50 tons/acre for many soils) often lead to a sealing of the soil surface, thereby impeding O_2 movement into the profile. The result is rapid O_2 depletion and equally rapid lowering of Eh to an undesirable range where it may remain for extended periods.

6. In view of energy shortages for agriculture and increased costs as well as probable deficits of fertilizer N, research should be initiated which asks the question, "How much waste of a particular N content should be applied to land in conjunction with other N inputs and the N requirement of the crop?"

7. We must begin to think in terms of acceptable loading rates rather than maximum loading rates; that is, loading rates that are compatible with specific wastes, soils, management practices, and which consider the N content of the waste relative to the N requirements of the crop.

8. The mode or method of application of a waste or residue on land will directly influence its' rate and extent of decomposition. Wastes that are soil incorporated would, under most conditions, decompose more rapidly than surface-applied or semi-incorporated wastes. However, there is some indication that some wastes (e.g., fresh animal manures) applied to soil in slug applications (i.e., in trenches or other localized placements) may actually decompose more rapidly than when thoroughly mixed with the soil. This rather unexpected phenomenon can probably be attributed to the nature and activity of the indigenous microflora associated with the waste. There is need for additional research in this particular area.

9. The rate and extent of decomposition of wastes and residues would be more rapid in neutral to alkaline soils than in very acid soils.

10. At a given loading rate high BOD wastes would tend to deplete the available soil O_2 and induce soil anaerobiosis more rapidly than low BOD wastes

11. Poorly drained soils of high clay content would undergo more rapid soil anaerobiosis from waste applications than lighter textured soils (i.e., sandy loams and silt loams) with good drainage. Extremely sandy soils would have a higher aeration potential but would be associated with nitrate leaching problems.

12. Loading rates and application frequencies may have to be balanced to maintain adequate soil O_2 levels for sustained aerobic decomposition and to prevent extensive and prolonged soil reduction.

REFERENCES

- Alexander, M. 1961. Organic matter decomposition, Chapt. 9. In Soil microbiology, p. 139-162. John Wiley & Sons, Inc., New York.
- International Symposium on "Livestock waste management and pollution abatement" held at Columbus, Ohio, April 19-22, 1971. Proceedings publ. by American Society of Agricultural Engineers, St. Joseph, Michigan, 360 pps.
- Mathers, A. C. and B. A. Stewart. 1970. Nitrogen transformations and plant growth as affected by applying large amounts of cattle feedlot wastes to soil. In Relationship of agriculture to soil and water pollution, pp. 207-214. Cornell Univ. Conf. on Agr. Waste Mgmt. Proc., Rochester, New York.
- McCalla, T.M., L.R. Frederick, and G.L. Palmer. 1970. Manure decomposition and fate of breakdown products in soil, Chapt. 17. In Agricultural practices and water quality, pp. 241-255. T. L. Willrich and G.E. Smith [Eds.]. Iowa State University Press, Ames, Iowa.
- McCalla, T.M. and F.A. Haskins. 1964. Phytotoxic substances from soil microorganisms and crop residues. Bacteriol. Rev. 28:181-207.
- McKinney, R.E. 1970. Manure transformations and fate of decomposition products in water, Chapt. 18. In Agricultural practices and water quality, pp. 256-264. T.L. Willrich and G.E. Smith [Eds.]. Iowa State University Press, Ames, Iowa.
- Parr, J.F. 1969. Nature and significance of inorganic transformations in tile-drained soils. Soils and Fertilizers 32:411-415.
- Peterson, J.R., T.M. McCalla, and G.E. Smith. 1971. Human and animal wastes as fertilizers, Chapt. 18. In Fertilizer technology and use, 2nd Edition, 671 p. R.A. Olson, T.J. Army, J.J. Hanway, and V.J. Kilmer [Eds.]. Soil Science Society of America, Madison, Wisconsin.
- Wadleigh, C.H. 1968. Wastes in relation to agriculture and forestry. U.S. Dept. Agric., Misc. Publ. No. 1065. 112 pp.
- Waksman, S.A. 1938. Humus. 2nd Edition. Publ. by The Williams and Wilkins Co., Baltimore, Maryland.
- Wilson, C.E. and F.E. Beckett [Eds.]. 1968. Proceedings of a Symposium, "Municipal sewage effluent for irrigation," publ. by Louisiana Tech. Univ. Alumni Foundation, Ruston, Louisiana.

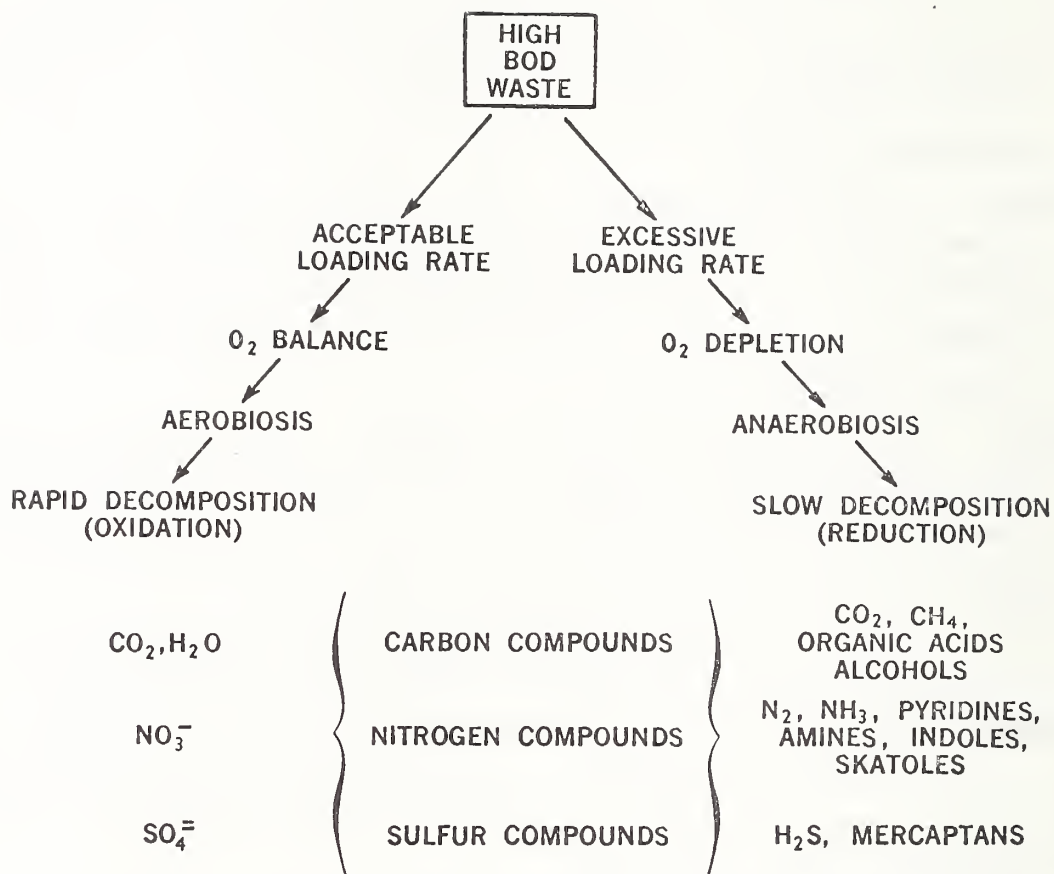


Figure 1. A flow diagram illustrating the possible consequences which might develop from the application of a high BOD waste to soil at an excessive loading rate compared with an acceptable loading rate.

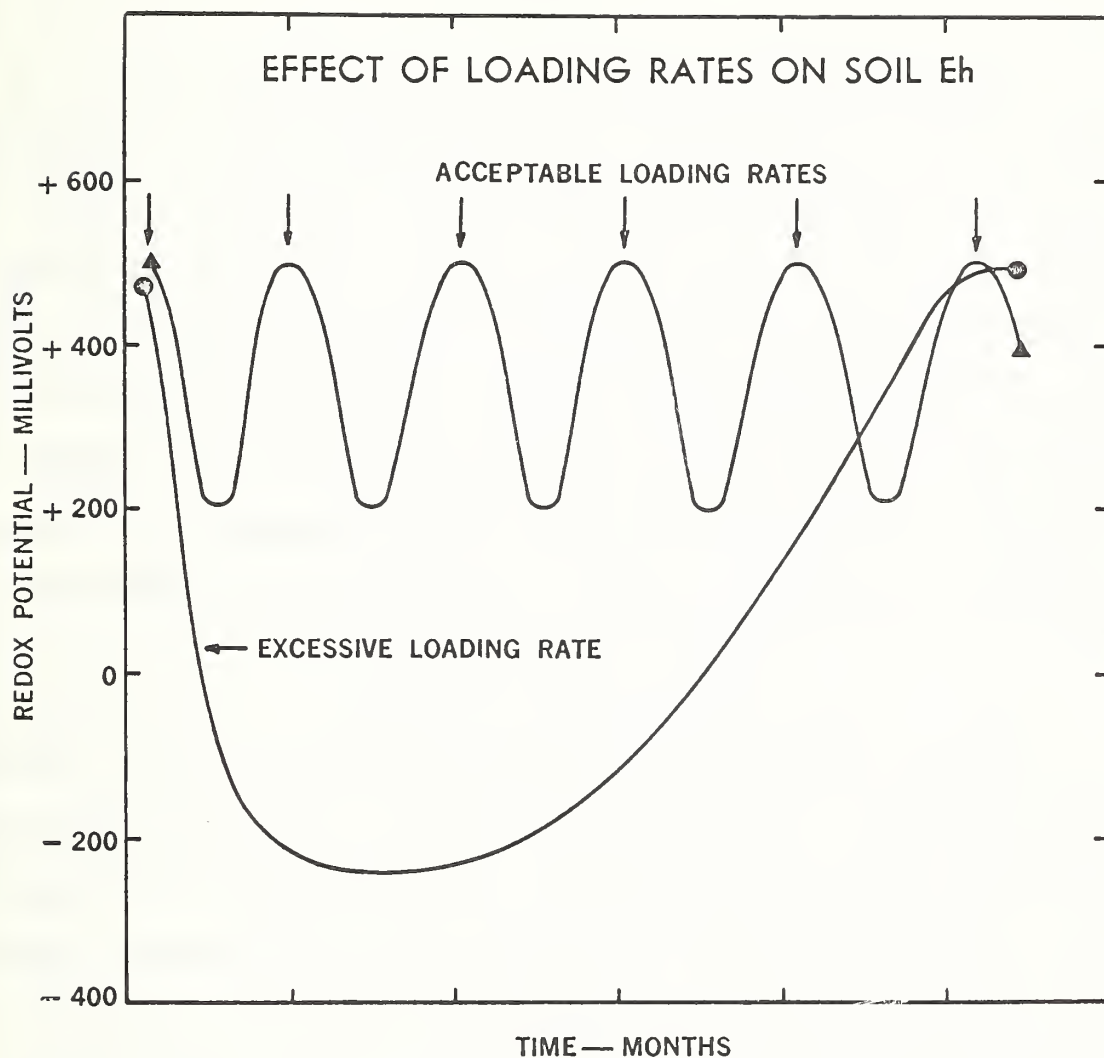


Figure 2. A hypothetical illustration of the relative effects of frequent, successive waste applications at acceptable loading rates compared with a single slug application on changes in the soil redox potential.

SALINITY PROBLEMS ASSOCIATED WITH WASTES^{1/}B. A. Stewart^{2/}

Wastes vary greatly in salt content and rates of waste applied to cropland cover a vast range. Also, textures of soils to which wastes are applied vary from sand to clay, climatic conditions under which they are accumulated and applied range from humid to arid, and wastes are applied to both nonirrigated and irrigated soils. Consequently, there is no simple way of indicating waste handling practices in which salinity problems will be avoided. The basic principles of salinity management apply, however, and before considering specific wastes and their management, it is necessary to be familiar with some of the basic terminology and principles of salinity.

Determining Soil Salinity

Salinity is generally determined by measuring the electrical conductivity of solutions. Salt molecules, in solution, split up to produce ions. The ions can conduct electrical current. The greater the concentration of ions in a solution, the greater the conductivity of the solution. Conductivity is measured in millimhos per centimeter (mmhos/cm). A solution containing 640 ppm of table salt has an electrical conductivity (EC) of

^{1/} Contribution from the Soil, Water, and Air Sciences, Southern Region, Agricultural Research Service, USDA, in cooperation with The Texas Agricultural Experiment Station, Texas A&M University.

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1 mmho/cm. This concentration is equal to 1 pound of salt in 200 gallons or 350 pounds in 1 acre-inch of water.

Soil salinity is generally appraised by measuring the electrical conductivity of the saturation extract. Because the saturation percentage is directly related to the field moisture range, conductivity of the saturation extract can be related to plant response. Under field conditions, the soil water content fluctuates between a lower limit represented by the wilting percentage and an upper limit represented by the field capacity percentage. The field capacity percentage is the amount of water that the soil can hold under conditions of free drainage after excess water has drained away following a rain or irrigation that has wet the whole soil. Measurements on soils of various texture show that saturation percentage is approximately two times field capacity, which in turn is about two times wilting percentage. Consequently, the soluble salt concentration of the saturation extract is about one-half that of the soil solution at the upper end of the field moisture range and about one-fourth of that at the lower, dry portion of the moisture range. For this reason, the conductivity of the saturation extract can be used to predict the effect of soil salinity on plant growth. The salt dilution that occurs in fine-textured soils, because of their higher water holding capacities, is automatically taken into account.

The range of conductance values generally found in soils and the effect of texture on conductance are shown in Figure 1. The line for a saturation percentage of 50 (water percentage on a dry weight basis) represents a fine-textured soil having a field capacity value of about

25 percent. The 25 percent saturation percentage line typifies a coarse-textured soil. The figure shows that a fine-textured soil containing 0.2 percent salt (equivalent to about 4,000 pounds in 6 inches of soil) would have a saturation extract conductivity value of 6 mmhos/cm. The same amount of salt in a sandy soil would result in a conductivity value of over 12 mmhos/cm. This illustrates the importance of soil texture in salinity management. However, even though salt additions influence sandy soils more than clay soils, salts are also more easily leached from sandy soils.

Salt Accumulation in Soils

Several factors affect the degree to which salts accumulate in a soil where wastes are added. It is necessary to take into account native salt present in the soil, if any, and the salt content of the waste added. Since the amount of salt taken up by a growing crop is negligible, salts may be either leached out of the soil profile or they tend to accumulate in the surface. The processes of evaporation from the soil surface and transpiration from plants remove only water from the soil, thereby concentrating the salts left behind.

Successive additions of solid wastes containing significant amounts of salts may result in detrimental accumulations of salts in the surface unless they are periodically leached out of the soil profile. In humid areas, this is generally accomplished by rainfall on well-drained soils. In semiarid and arid areas, irrigation water in excess of crop requirement must be applied to leach the salts.

When applying sewage effluents, processing plant effluents and water from animal waste lagoons to the land, large volumes of water are involved. Salt accumulations can generally be avoided by controlling application rates to provide the required leaching. By taking into account the salt content of the waste applied, climatic conditions, soil characteristics and plant use, it should be possible to adequately control salinity and maintain good vegetative cover.

Salinity Effect on Plants

A generalized index of the effect of soil salinity on plant growth is shown in Table 1. Plants that are salt sensitive or only moderately tolerant show progressive declines in growth and yield as levels of salinity increase. Figures 2, 3, and 4 show salt tolerances of some important crop plants. Indicated tolerances apply to the period of rapid plant growth and maturation, from the late seedling stage onward. These data show that crops like barley and bermudagrass can tolerate very high salinity levels, while beans and clover are severely affected by small amounts of soluble salts.

In selecting crops, particular attention should be given to the effect of salinity during germination. Poor crops often result because of poor stands. This problem is complicated by the fact that some crop species that are very salt tolerant during later stages of growth may be sensitive to salinity during germination. Modification of seeding practices can minimize salt accumulation near the seed. Some of these practices will be discussed later.

Sodium Hazard

The concentration of specific ions, as well as the total soluble salts, is an important consideration. Sodium is very common under saline conditions and can affect crop production in many ways. A common measurement made on soil extracts and irrigation waters is the sodium-adsorption-ratio (SAR). This ratio expresses the degree to which sodium will be adsorbed by soils from a solution in equilibrium with the soil. Specifically the $SAR = \frac{Na^+}{\sqrt{(Ca^{++} + Mg^{++})/2}}$, where the ionic concentrations are expressed in milliequivalents per liter.

Sodium toxicity may become a factor for sodium sensitive fruit crops such as avocados, stone-fruits, and citrus when the SAR reaches about 4. For most crops, the restrictive factor is the poor soil physical condition caused by sodium. When calcium is the predominant cation adsorbed by the soil, the soil tends to have a granular structure and is easily worked and readily permeable. As sodium replaces calcium, the soil particles become dispersed and poor physical properties result. For general crops, a limit of 8 to 18 for the SAR is considered within an acceptable range. This range depends to some degree on the type of clay mineral predominant in the soil, the soluble salt concentration, and other variables. Sodium-adsorption-ratios can be lowered by supplying calcium by soil amendments such as gypsum, or by treating a soil containing unsoluble lime with acid or acid-forming amendments to release calcium.

Salinity Considerations of Wastes

Farm animals excrete large amounts of salts. Table 2 gives the approximate amounts of salts excreted by various animals. Table 3 shows

the concentration of dissolved solids and electrical conductivity of various wastes commonly applied to land. Table 4 gives the concentration of selected elements for wastes commonly applied to cropland. These examples can be useful in considering potential salinity resulting from applying wastes to cropland.

Beef Cattle Wastes

The salinity level of beef cattle waste can vary greatly depending on the ration, climatic conditions, and manure handling methods. However, all beef cattle wastes, even those comparatively low in salt concentrations, contain relatively large amounts of soluble salts. Potassium, sodium, calcium, and magnesium contents of manure are of primary importance when considering salinity. These cations are generally present in manure as soluble salts with chloride and sulfate as the most common associated anions. The data in Table 4 show potassium levels ranging from a low of 1.02 to a high of 3.64 percent. Similar variations are shown for calcium, sodium, and magnesium. A primary reason for these large differences is probably climatic variations. Manure from feedlots in dry areas (Bushland, Texas, and Brawley, California) are much higher in salts than manure from feedlots in the wetter areas (Pratt, Kansas, and Mead, Nebraska). In dry areas, there is much less runoff from the lots and this results in smaller amounts of soluble salts being washed from the manure. Also, stocking rates are much lower in wet areas and this results in the salts being dispersed over larger areas where they are more susceptible to leaching and runoff.

In view of the large variations in the salt concentrations of manure, chemical analyses are advised. This allows an estimate of salinity. For

example, for every 6 tons of dry manure (approximately the amount applied in 10 tons of wet manure from a stockpile) from Brawley, California, (Table 4), applied to an acre of cropland, about 437 pounds of potassium, 161 pounds of sodium, 164 pounds of calcium, and 96 pounds of magnesium would be added to the soil. The chlorides and sulfates accompanying these cations weigh even more. Consequently, about 2,000 pounds of soluble salts would be added. In contrast, only about 500 pounds of salts would be applied to soil treated with the same amount of manure from Mead, Nebraska.

A 6-inch depth of soil weighs about 2,000,000 pounds. Therefore, if 2,000 pounds of soluble salts were applied with manure and this was incorporated in the surface 6 inches of soil, the salt content would be increased by 0.1 percent. Referring to Figure 1, this amount of salt could result in a saturation extract conductivity value of about 3 mmhos/cm when applied to a fine-textured soil, and about 6 mmhos/cm for a coarse-textured soil. This is an extreme example because it is based on manure very high in soluble salts and it is assumed that the salts remain in the surface 6 inches of soil. Generally, much of the salt will be moved to deeper depths of the soil profile. However, it illustrates the importance of applying manure well in advance of planting time to allow time for leaching. It should also be remembered that large applications of manure may result in salinity hazards, even when the manure is relatively low in soluble salts. An application rate of 60 tons per acre of dry manure from Mead, Nebraska, would add about 5,000 pounds of soluble salts per acre.

Water management plays a dominant role in determining the effect of manure applications on salinity buildup in soil. Leaching of salts from

soil profiles depends on a number of factors. The most important ones are infiltration rate, climate (evapotranspiration), frequency and amount of irrigation, and the cropping systems. Several small irrigations are generally more effective in leaching salts than one large irrigation. If manure is applied to land under conditions of high rainfall or large amounts of irrigation water are applied, soluble salts will be dispersed through a large portion of the profile or leached below the root zone. In contrast, if manure is applied to cropland under water deficient conditions, salt concentrations can build up. Even under these conditions, management practices can be altered to minimize problems. When salt concentrates in soil, germination of seeds often becomes the most serious problem. The salt concentration builds up in the surface inch or so of the soil because of upward movement of the soil solution and surface evaporation. The germinating seed may be exposed to salt concentrations several times greater than those encountered by the roots at later growth stages. This condition is often present in furrow-irrigated row crops. Water, moving from the furrows into the ridges, moves soluble salts into the center and crown of the bed. Salt concentrations in centers of beds are commonly 5 to 10 times greater than original salt concentrations of the plow layer. Figure 5 shows areas of salt accumulation in beds with different planting practices. Planting seed on sloping beds can greatly minimize salinity hazards because the advancing water front sweeps salt past the seed instead of up to it.

The application of manure may also result in sufficiently high sodium concentrations in soils to degrade soil physical properties. However, the data in Table 4 indicate that this should not be a serious problem under

most conditions since there is a fairly good balance between sodium, and calcium and magnesium, which tends to keep the sodium-adsorption-ratio from becoming too great. Nevertheless, the sodium status of the soil should be carefully followed and every attempt made to keep the sodium content of manure as low as possible.

Poultry Manure

The element composition of poultry manure is somewhat similar to that of beef cattle manure (Table 4). The main difference is that there is less sodium in relation to calcium. This is desirable from the standpoint of reducing the sodium hazard. Also, the nitrogen content of poultry manure is substantially higher than that of beef cattle manure which results in generally smaller amounts of poultry manure being applied to cropland.

Sewage Effluent and Sludge

The inorganic constituents of raw sewage are largely determined by the characteristics of the carriage water. Consequently, wide variations in inorganic concentrations occur. The inorganic ion concentration in domestic sewage averages about 500 ppm with a range of about 300 to 3,000 ppm. Table 4 shows the distribution of principle cations in sludge and effluents. Effluent containing 500 ppm of inorganic ions has an electrical conductivity value of about 0.75 and is very suitable for application on cropland. With careful management sewage water containing as much as 3,000 ppm of inorganic constituents may be used on cropland. This water would have an electrical conductivity value of about 4 mmhos/cm. Where water is to be used regularly for irrigation of relatively impervious soil,

its value is limited if it contains more than 2,000 ppm of dissolved inorganic salts. This water would have an electrical conductivity of about 3 mmhos/cm.

Application of sludge to cropland usually does not cause salinity problems. The readily soluble salts have been removed with the effluent. Also, the material remaining as sludge has a high ratio of calcium to sodium.

Miscellaneous Wastes

Cropland is increasingly being used as an acceptor of wastes from many sources. These wastes are applied both as solids and as liquids. The salinity status of these materials should be closely monitored to prevent problems from developing. When applied as liquid, the electrical conductivity of the solution should generally be kept below 2 mmhos/cm and the sodium-adsorption-ratio below about 10. In many cases, wastes can be diluted with other water sources to obtain suitable effluent to dispose on land. This is often the case of impounded runoff from feedlots. The electrical conductivity of feedlot runoff is often in the range of 6 to 9 mmhos/cm, too high for application to many crops. Consequently, some dilution with other water is often necessary. When solids are applied to land, an analysis of the material is essential to assessment of potential salinity problems. The basic principles of soil salinity are well established and can be applied to essentially all wastes.

SUMMARY

Applications of wastes to cropland can result in salinity problems. However, serious problems can usually be averted by good management.

Salinity problems are most likely to occur when land is used as a "dumping ground" for wastes and exceedingly large rates are applied. Many wastes commonly applied to land contain substantial quantities of nitrogen. Generally, if application rates are limited to quantities necessary to supply adequate nitrogen for crop production, salinity problems will not be serious. This is particularly true for beef cattle manure and poultry manure. Excess nitrogen problems will usually result sooner than salinity problems.

The most important consideration in applying wastes to land is the rate of application. Problems from utilizing wastes on land are usually caused by applying too much waste. When proper amounts are added, the soil is an ideal acceptor, the waste is degraded, and the soil is generally improved.

REFERENCES

- Bernstein, Leon. 1964. Salt tolerance of plants. Agriculture Information Bulletin No. 283. USDA.
- U. S. Cold Regions Research and Engineering Laboratory. 1972. Wastewater management by disposal on the land. Special Report No. 171. Corps of Engineers, U.S. Army.
- United States Salinity Laboratory Staff. 1954. Diagnosis and improvement of saline and alkali soils. U.S. Department of Agriculture Handbook No. 60.

Table 1. Responses of crops grown in soils of varying electrical conductivity values of the soil saturation extracts (EC).

Salinity EC mmhos/cm at 25° C	Crop responses
0- 2	Salinity effects mostly negligible.
2- 4	Yields of very sensitive crops may be restricted.
4- 8	Yields of many crops restricted.
8-16	Only tolerant crops yield satisfactorily.
Above 16	Only a few very tolerant crops yield satisfactorily.

Table 2. Approximate amount of salts excreted by farm animals^{1/} (From Branson and Pratt, University of California at Davis).

Type of animal	Number	Total potential salts
		<u>Lb/yr</u>
Dairy cows ^{2/}	1	1,318
Beef steer ^{2/}	1	618
Calf	1	365
Poultry		
Broilers	1,000	5,744
Layers	1,000	13,248
Turkeys	1,000	19,721

^{1/} The values depend on animal size, the ration, and to a small extent the salt in drinking water. The amount of salt retained in soils will be of the order of magnitude of one-half that excreted.

^{2/} Producing dairy cows of 1,300-lb weight, steers of 500- to 1,000-lb weight, and calves of 250- to 500-lb weight. The higher value for calves compared to steers is because of differences in rations comparing calves before the feedlot and steers in the feedlot.

Table 3. Some values for total dissolved solids in various wastes.^{1/}

Source	TDS	EC
	<u>Ppm</u>	<u>Mmhos/cm</u>
Municipal wastewater	760-2,000	1.2-3.1
Processing plant effluents ^{2/}	200-3,000	0.3-4.7
Beef cattle waste ^{3/}	200-2,500	0.3-3.9
Feedlot lagoon water	8,300	13.0
Sewage sludge	2,500-4,000	3.9-6.1

^{1/} Data represent a range of samples where data were available. Ranges presented do not represent actual limits.

^{2/} Does not include peeling and processing potatoes where TDS may run as high as 7,000 to 10,000 ppm.

^{3/} Depends on degree of freshness and presence of extraneous material.

Table 4-- Composition of selected wastes commonly applied to cropland.

	N	P	K	Na	Ca	Mg	Fe	Zn
	Percent on dry solids							ppm
<u>Beef cattle manure</u>								
Mead, Nebraska	--	--	1.02	0.17	0.48	0.31	0.99	62
Pratt, Kansas	1.59	0.54	1.40	0.32	1.03	0.50	1.02	102
Bushland, Texas	2.05	0.81	2.29	1.13	1.98	0.76	0.32	137
Brawley, California	3.12	0.91	3.64	1.34	1.37	0.80	0.43	110
<u>Poultry manure</u>	3.80	1.45	1.91	0.39	1.91	0.47	0.09	286
<u>Dried sewage sludge</u>	1.07	0.34	0.74	0.39	3.43	0.64	2.37	1460
<u>Digested municipal sludge</u>	5.50	--	0.41	0.47	4.30	1.20	--	--
	Parts per million							
<u>Sewage effluent</u>	20	9	9	70	20	5	--	--

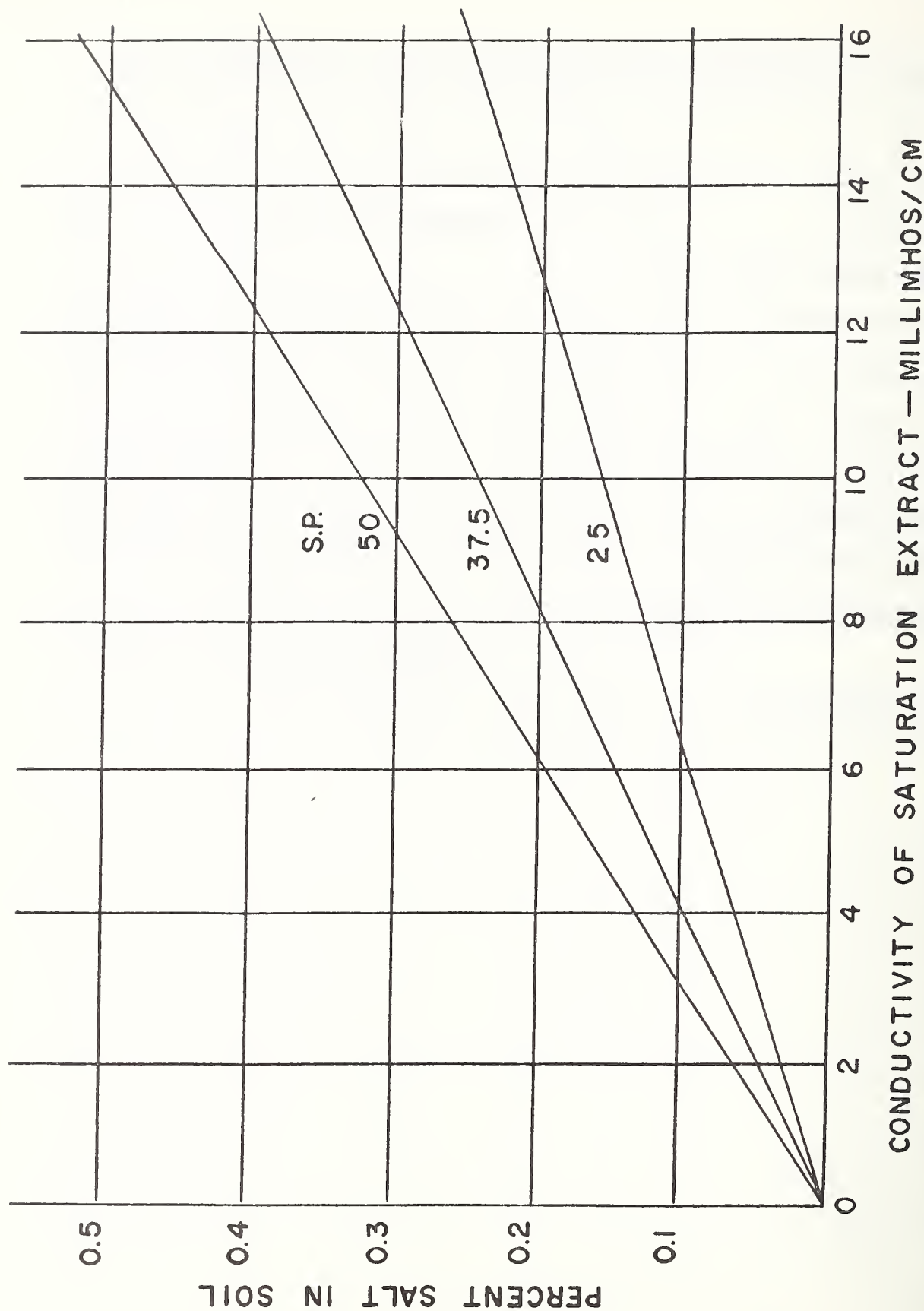


Figure 1. Range of conductance values found in saturation extracts of soils as influenced by texture and salt content. [Saturation percentages (S.P.) of 50 and 25 represent heavy-textured and light-textured soils, respectively.]

SALT TOLERANCE OF FIELD CROPS*

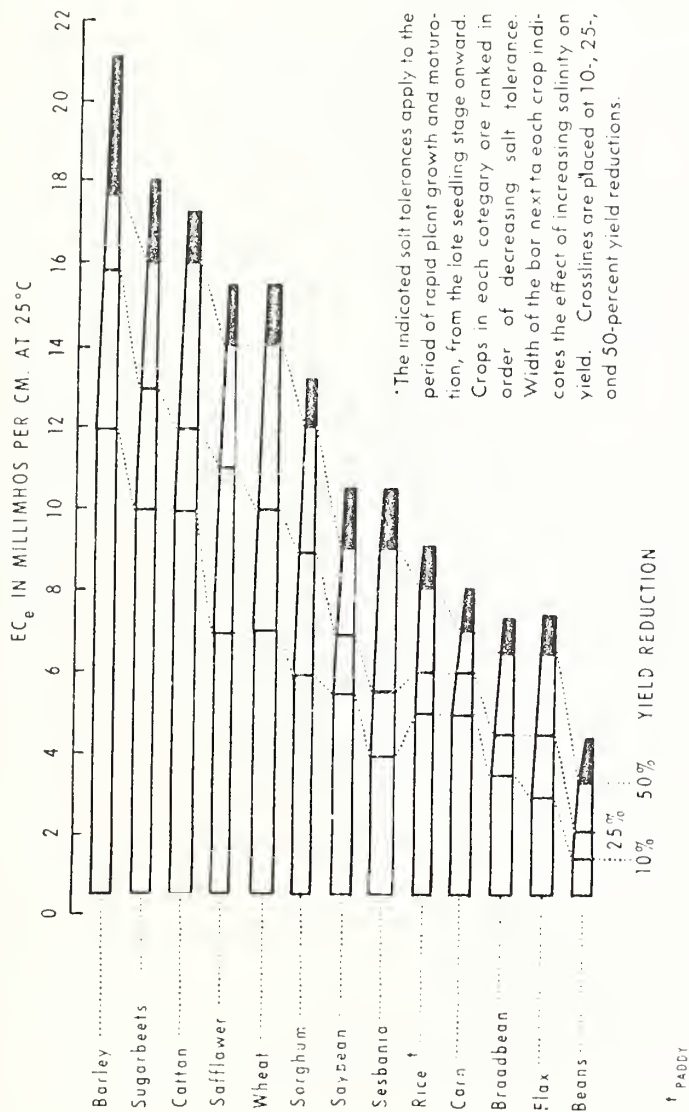


Figure 2. Effect of soil salinity on growth of field crops. (From Bernstein, L. 1964.

Salt tolerance of plants. USDA Agr. Inf. Bull. No. 283. 24 p.)

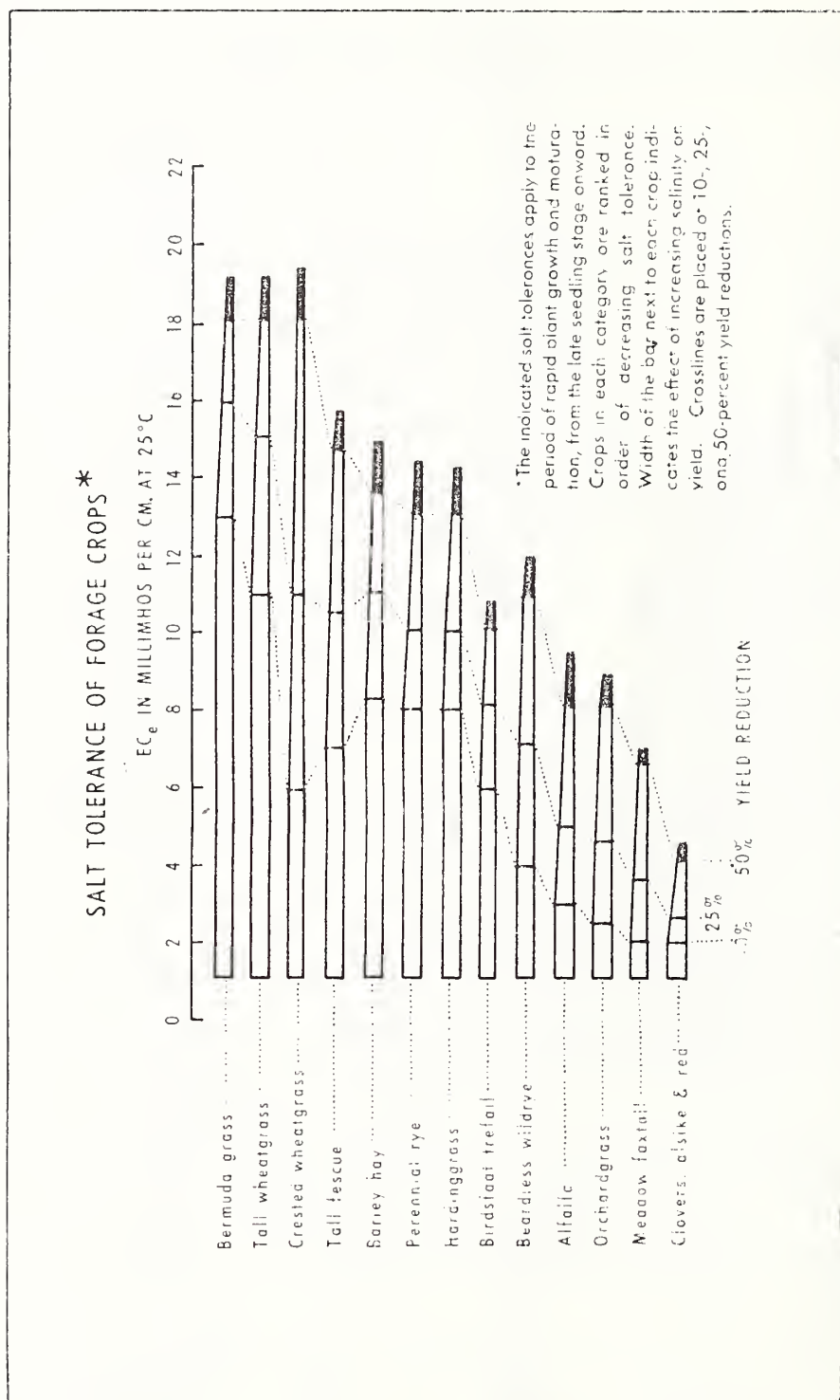
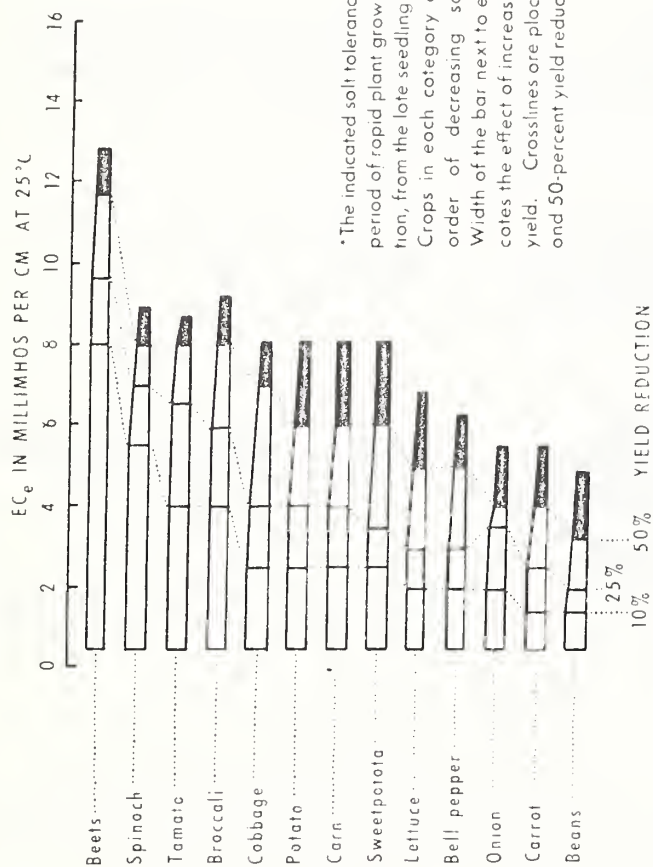


Figure 3. Effect of soil salinity on growth of forage crops. (From Bernstein, L. 1964.

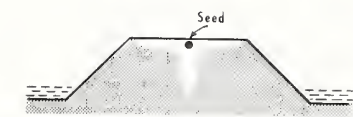
Salt tolerance of plants. USDA Agr. Inf. Bull. No. 283. 24 p.)

SALT TOLERANCE OF VEGETABLE CROPS *



*The indicated salt tolerances apply to the period of rapid plant growth and maturation, from the late seedling stage onward. Crops in each category are ranked in order of decreasing salt tolerance. Width of the bar next to each crop indicates the effect of increasing salinity on yield. Crosslines are placed at 10-, 25-, and 50-percent yield reductions.

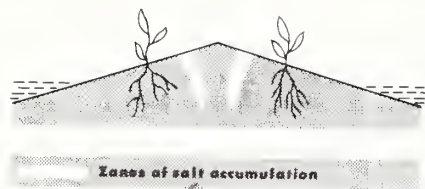
Figure 4. Effect of soil salinity on growth of vegetable crops. (From Bernstein, L. 1964. Salt tolerance of plants. USDA Agr. Inf. Bull. No. 283. 24 p.)



Single-row bed. Germination failures are frequent, even if soil is only slightly saline at time of planting.



Double-row bed. Germination is affected at moderate salinity levels.



Sloping bed. Germination is rarely affected by salt initially present in soil.

Diagrams showing areas of salt accumulation in raised beds with different planting practices. 8\ - 8381

Figure 5.-- Diagrams showing areas of salt accumulation in raised beds with different planting practices. (From Bernstein, L. 1964. Salt tolerance of plants. USDA Agr. Inf. Bull. No. 283. 24 p)

HYDRAULIC ASPECTS OF LIQUID-WASTE LOADINGS 1/Herman Bower 2/Types of Systems

Land application systems for the disposal and treatment of liquid waste (sewage effluent, processing plant wastes, lagoon effluents, etc.) can be divided into low-rate systems and high-rate systems. With the low-rate systems, the waste water is applied in small doses, for example a fraction of an inch per day (applied in a few hours or so), a few inches per week (applied in one day), etc. With such systems, the application rates and amounts are usually well below the infiltration capacity of the soil. An exception may be very heavy soils, soils with a plow pan or other restricting layer at some depth, or poorly drained soils, but then such soils would not be very suitable for liquid waste disposal systems in the first place. While low-rate systems are preferred because of minimum environmental impact, the large land requirements can be a disadvantage, especially if appreciable volumes of waste water need to be disposed. For example, at 2 inches per week application, a disposal area of 130 acres would be required per one mgd of effluent.

With high-rate systems, usually as much effluent is applied as the soil can absorb. As with low-rate systems, infiltration periods

1/ Contribution from the Agricultural Research Service, U. S.

Department of Agriculture.

2/ Director, U. S. Water Conservation Laboratory, 4331 East Broadway
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are rotated with dry periods to allow oxygen to enter the soil and to restore infiltration rates. Permeable soils should be selected for high-rate systems. Application amounts may range from several feet to several yards per week, depending on the hydraulic conductivity of the soil and the build-up of groundwater mounds as determined by the aquifer characteristics. While sprinklers are generally used for low-rate systems, basins or furrows can be used for high-rate systems if the topography is favorable.

Since the nitrogen load on the soil with high-rate systems may be much greater than can be removed from the soil by growing a crop, the application system and application schedule must be selected so as to maximize denitrification in the soil. Favorable conditions for denitrification are more easily created with basins or furrows, where low oxygen levels in the soil can be created simply by extended flooding. Use of sprinklers, involving drainage between sprinkler revolutions and resulting entry of air into the soil, may cause predominantly aerobic conditions, particularly on coarse, rapidly draining soils. Such aerobic conditions will result in essentially complete conversion of the nitrogen in the waste water to nitrate, and also in essentially complete oxidation of organic carbon (unless the BOD of the waste water is very high). Thus, subsequent denitrification is not possible and the nitrogen stays in the water as nitrate.

Where soils of sufficient permeability for even low-rate systems are not available, overland flow whereby the waste water flows over

vegetated soil in a shallow sheet, is sometimes used. These systems may consist of vegetated borders if the land is flat. On rolling land, the water may be applied by a sprinkler line on the contour and collected by a terrace at a certain distance downhill from the sprinkler line. Such overland flow or "vegetation filtration" systems can efficiently remove suspended solids and BOD, and in some cases also considerable amounts of nitrogen and phosphorus. The design of overland flow systems is mainly based on providing sufficient surface detention time to obtain a runoff of desired quality.

Regardless of whether a low-rate or high-rate system is selected, field investigations should be carried out prior to constructing the system, particularly if there is some question whether or not the soil can handle the desired application.

Predicting Infiltration Rates

The prediction of infiltration rates for disposal fields is more difficult than the prediction of "normal" infiltration rates in watershed hydrology or irrigation design where rainfall or fairly high-quality water is used. With land application of waste water, however, the infiltrating water may contain suspended solids that accumulate on the surface of the soil, and organic compounds that stimulate bacterial action in and on the soil. The cations in the waste water could have an unfavorable effect on soil structure. These actions all decrease the infiltration rates. The best way to determine the attainable infiltration rates for a land application system is through field

trials, using the actual waste water and application system to be employed (sprinklers, furrows, or basins). These trials may also serve to evaluate optimum amounts and frequencies of application and to determine the effectiveness of vegetation or other surface treatments of the disposal field.

Sometimes, infiltration measurements with clean water may be desirable to determine the "potential" hydraulic loading rate of a field and to compare fields as to their suitability for wastewater disposal. The usual precautions should be taken to make sure that the results are applicable to the entire field. Thus, cylinder infiltrometers should be of sufficiently large diameter (at least 3 feet) to minimize the effect of flow divergence on the measured infiltration rate. Also, the infiltrometer (cylinder or sprinkler) should cover a sufficient area to give realistic infiltration rates in case restricting layers are present at some depth in the soil profile. When a small infiltrometer is used under those conditions, lateral flow will occur in the groundwater mound that will be formed above the restricting layer. The infiltration inside the cylinder will then be higher than the infiltration rate for a large area, where lateral flow above the restricting layer cannot occur.

Care should be taken when using infiltrometers on sloping land with shallow soil. In such cases, the infiltration capacity of the field may be determined by how much water can flow downhill as subsurface runoff in the soil above the restricting layer. If water

is applied to a small area, as with an infiltrometer, all the water that infiltrates could easily move downhill as subsurface runoff. However, if larger areas are wetted, the subsurface runoff may not be sufficient to dispose of all the infiltrated water. In that case, the soil becomes saturated and surface runoff will occur. The actual infiltration rate is then only a fraction of that indicated by a small infiltrometer.

Where restricting or impermeable layers are present in the soil, it may be better to predict the potential infiltration rates from measurements of the hydraulic conductivity profile of the soil. Techniques are available for field measurement of hydraulic conductivity at different depths, above as well as below a water table. The final infiltration rate can then be calculated by applying Darcy's equation to the one-dimensional flow system. If the field is sloping and the applied waste water can only move away laterally as subsurface runoff, the potential subsurface flow can be computed. This value can then be divided by the proposed width of the disposal field to obtain an estimate of the safe application rate.

The actual average long-term infiltration rate or "hydraulic loading" for waste water will be less than the potential rates because of clogging by suspended solids and biological action, and the need for drying or resting periods. Also, the ionic composition of the waste water may cause dispersion of the clay in the soil with a resultant decrease in hydraulic conductivity. The relationship between

potential and actual infiltration should be developed for various wastes, soils, and climatic conditions, so that potential infiltration rates can be converted into design application rates for waste-water disposal systems. For a high-rate system near Phoenix, Arizona, for example, secondary sewage effluent is infiltrated from basins at a water depth of 1 foot using schedules of 2 to 3 weeks flooding alternated with 10 to 20 days drying. The long-term hydraulic loading was found to be about 23% of the potential infiltration rate (i.e., 350 ft/yr versus 1,500 ft/yr). Similar data should be obtained for other proposed systems.

Soil Clogging

Land disposal of liquid waste invariably causes clogging of the soil and resulting decline in infiltration rates. Clogging occurs on the surface, when suspended solids do not move into the soil. Fine, suspended matter may actually move some distance into the soil, particularly in coarse textured soils. Microorganisms growing on the soil surface and in the soil pores may also contribute to a permeability decrease because of the space they occupy and the products they form. High dissolved BOD-values may cause serious clogging problems because of the bacterial activity they stimulate.

If clogging of the soil surface is caused by an inert material, the impedance of the clogged layer can be expected to increase linearly with the total solids load, assuming that the clogged layer builds up uniformly. Since the clogged layer may become more compact as

the total solids load increases, however, the impedance of the clogged layer may actually increase more rapidly than the total solids load. This is particularly true if the hydraulic gradients at the soil surface are relatively high. The clogged layer is then subjected to a seepage force which causes compaction of this layer.

An unfavorable situation occurs where coarse soil overlies finer soil, or where the soil is covered by a gravel layer to provide "better" infiltration. In that case, particulate matter may move through the coarse upper layer and settle out on the finer soil layers further down. The clogging caused by this is out of reach of the drying influence of the sun or of the mechanical influence of cultivation equipment. Thus, it may take a very long time of "resting" the system before deep accumulations of organic clogging materials are decomposed and infiltration rates are restored.

The depth and extent of the clogged zone can be evaluated with tensiometers installed in the soil at different depths. Soil clogging occurs where the hydraulic gradients increase with time. A quick way of assessing where clogging occurs is to flood an area of soil and measure the effect of water depth on infiltration rate. Applying Darcy's equation to the infiltration system shows that if the infiltration rate increases essentially linearly with the water depth, the clogged layer is thin and at the surface of the soil. However, if the infiltration rate increases less than linearly with increasing water depth, clogging takes place at greater depths or over a greater distance in the soil profile. Surface clogging is the easiest to control

or prevent. Drying, harrowing, scraping, etc. are some of the practices that have been effectively used to reduce the hydraulic impedance of the clogged layer. Of course the best way to control clogging is to prevent it by minimizing the suspended solids content of the waste water, for example, by using settling basins, and if needed, flocculants and/or filtration.

Scheduling Infiltration and Dry Periods

With continued application of waste water, organic clogging causes the infiltration rates to eventually become so low that a drying or resting period is necessary to restore the infiltration rates. Such drying will allow dessication and decomposition of the organic clogging material, which is usually effective to give complete recovery of the infiltration rate. Occasionally, some mechanical treatment such as scraping the surface to remove accumulated solids, disking, or raking, or an extra long dry period may be necessary. Inorganic clogging materials, such as silt and other fine sediment, should be removed from the waste water before applying it to the land.

If it is desirable to minimize the land area required for the disposal system, the optimum combination of infiltration and drying periods should be evaluated. The application of waste water should be stopped before the infiltration rates become too low, and it should not be resumed until the infiltration rates have made reasonable recovery. If the decrease in infiltration during application

and the recovery in infiltration during drying are known, the combination of application and resting periods yielding maximum long-term infiltration or hydraulic loading can be evaluated. Since the optimum combination of infiltration and dry periods depends so much on the climate, the waste, and the soil characteristics, it can best be evaluated by local experimentation.

Combinations used in actual practice range from a few hours infiltration every 24 hours (for processing wastes) to infiltration and drying periods of several weeks each (for sewage effluent and high-rate infiltration basins). Minimizing suspended solids in the waste water is one of the best techniques for maintaining high infiltration rates.

Role of Vegetation

Vegetation is generally desirable for land treatment systems because it removes nutrients and other constituents that were applied to the soil with the waste water. Vegetation may also be effective in stimulating denitrification in the root zone causing additional nitrogen removal from the soil. Vegetated soils also dry to greater depths and to a lower water content than non-vegetated soils during the growing season. This will increase the oxygen diffusion rate, and hence the amount of oxygen entering the soil and the depth of oxygen penetration, during drying.

Vegetative covers (Reed canarygrass, bermudagrass, etc.) for the infiltration fields are usually recommended where waste water is

applied with sprinklers. This protects the soil surface against the direct impact of the drops, which could otherwise have an adverse effect on the structure of the surface layer and, consequently, on the infiltration rate. Vegetation also intercepts some of the suspended solids, leaving less solids to accumulate on the soil surface. New growth of vegetation may also be effective in breaking the continuity of impeding layers on the soil surface, which in turn should have a beneficial effect on the infiltration rate. Vegetation also reduces soil erosion.

Where the waste water is applied in basins, a disadvantage of vegetation is that it restricts the water depth that can be used in the basin. This will in turn result in lower infiltration rates, particularly where there is surface clogging of the soil which causes the infiltration rate to vary almost linearly with water depth. On the other hand, the water depth in the basin may have to be small enough to allow enough light to penetrate to the bottom for algal growth. When algae grow on the bottom, pieces of the algal mat may actually float up when oxygen bubbles become entrapped in the mat. Since suspended solids are carried up with the algae, this process "rejuvenates" the bottom of the basin and can actually increase the infiltration rates during flooding. When vegetation is used in the basins, flooding and drying periods may have to be scheduled so as to allow the vegetation to develop and survive. The resulting flooding and drying schedules may not be the most desirable from a standpoint of maximizing infiltration and/or maximizing nitrogen removal by

denitrification. Thus, the use of vegetation in infiltration basins depends on whether infiltration or denitrification needs to be maximized, or whether agricultural utilization of the basins is desirable.

The use of gravel layers or similar mulches may increase the infiltration rates initially, but their effectiveness decreases with time, and once they are clogged with solids the infiltration rate is difficult to restore.

Water Table Response

In the long run, waste water can be applied to the soil at no higher rate than the internal drainage rate of the soil. Where a restricting layer is present at some depth and the initial infiltration rate is higher than the rate of water movement through this layer, a perched water table will form and rise above the restricting layer. This water table will continue to rise until the rate of water movement through the restricting layer equals the infiltration rate. If the hydraulic conductivity of the restricting layer is too small for this to be reached, the water table will continue to rise until it reaches the soil surface. The infiltration rate then becomes equal to the rate of water movement through the restricting layer. At this point a sharp decrease in infiltration will occur which will lead to surface runoff if the waste water is applied by sprinklers. The effects of perching mounds or restricted subsurface runoff should be considered when selecting safe hydraulic loading rates, as mentioned in the section on predicting infiltration rates.

Rising water tables also occur if the rate of movement of the water below the water table is smaller than the infiltration rate. This can occur if the hydraulic conductivity of the aquifer is fairly low or if the aquifer is relatively shallow and the groundwater moves essentially in horizontal direction. Thus, an analysis of the groundwater flow system below the disposal fields should be made to make sure that the aquifer can transmit the infiltrated water at a sufficient rate to avoid build-up of groundwater mounds.

If horizontal-flow theory is used in the analysis of the underground flow system, the effective transmissibility rather than the transmissibility of the complete aquifer should be used. This is because groundwater recharge flow systems have an active upper zone and a passive deeper zone. Thus, only the upper portion of the aquifer is utilized, causing the effective transmissibility to be less than the total transmissibility, especially for deep, unconfined aquifers.

A good way to evaluate hydraulic properties of aquifers for recharge is with a pilot project. The effective transmissibility can then be determined from the infiltration rate and the response of the water levels in two observation wells of different depths in the center of the project. The results can be used to predict the groundwater mound and flow system for a larger disposal system.

To ensure rapid drainage of the soil profile during drying, which is necessary for effective oxygenation of the soil, the groundwater mound should not be allowed to rise closer to the soil surface

than a distance of about 5 feet, unless the mound rapidly recedes after infiltration is stopped.

In soils with restricted drainage, such as shallow or slowly permeable soils, an artificial drainage system may be needed for rapid lowering of the water table when drying is started. Drainage systems may also be needed to prevent groundwater tables from rising to the surface and restricting infiltration rates while waste water is being applied. Such drainage systems can be designed in the same way as agricultural drainage systems.

Minimizing Ground-Water Contamination

After the infiltrated waste water has reached the groundwater, it usually spreads in a lateral direction and eventually moves in accordance with the natural flow pattern of the groundwater. Some mixing can be expected at the "renovated-water front," but once the front is passed the native groundwater will be essentially displaced by the renovated water. Dilution of the renovated water by rain water may be significant for low-rate systems in humid areas. The renovated waste water may eventually surface again if the groundwater drains into a stream or lake, or it may be pumped with the groundwater from wells.

While the quality of the waste water is considerably improved as it moves through the ground and becomes renovated water, it may still be not as good as the native groundwater. Thus, a large, uncontrolled spread of the renovated water into the groundwater

basin may be undesirable, particularly if groundwater is pumped for municipal and industrial uses. If not naturally draining to surface water, the underground spread of renovated water can be checked by intercepting it with artificial drains or pumped wells. After collecting the renovated water in this manner, it can be re-used (after additional treatment, if necessary) or it can be discharged into surface water.

The proper time and distance of underground travel to obtain a renovated water of suitable quality depend on the waste characteristics, the desired quality of the renovated water, and the soil type. The important thing is to avoid travel through fissured rock or cavernous limestones or dolomites since not much quality improvement can be expected there. Although most bacteria and viruses are removed after a few feet of percolation through granular soil, longer travel distances are usually desired to protect against break-through of pathogenic organisms and to produce a renovated water of optimum quality without taste and odor. For a number of high-rate systems, underground travel times of several weeks and underground travel distances of several hundred feet are allowed before the renovated water is collected.

Underground travel times can be evaluated for existing systems by using tracers. Also, estimates of travel times can be obtained by a theoretical analysis of the flow system to construct the pattern of streamlines and equipotential lines. From this pattern, the macroscopic velocity of the groundwater can be evaluated, which enables

the determination of the travel time between the point of infiltration of the waste water and the point where it is collected as renovated water. The theoretical approach can be used in the design of new systems to make sure that the underground detention times are acceptable.

Pilot Projects

The performance of a low-rate or high-rate system for applying waste water to soil is very much dependent on the local conditions of climate, topography, soil, and hydrogeology. Unless similar projects already exist in the area, it is very desirable to experiment with a small pilot project before going into any large-scale development. The results from the experimental project can then be used as input information for the design and management of the operational system. Experimentation with a pilot project could well save a lot of disappointment and money later on. Finally, the best designed system for applying waste water to land will still be a failure if it is not properly managed.

BIBLIOGRAPHY

1. Bouwer, Herman. Returning wastes to the land, a new role for agriculture. Jour. Soil and Water Conserv. 23, 164-168, 1968.
2. Bouwer, H. Planning and interpreting soil permeability measurements. Jour. Irrig. and Drain. Div., Proc. Amer. Soc. Civ. Engin., 95 No. IR3, 391-402, 1969.
3. Bouwer, H. Ground-water recharge design for renovating waste water. Jour. Sanit. Engin. Div., Proc. Amer. Soc. Civil Engin., 96 No. SA1, 59-74, 1970.
4. Bouwer, H., R. C. Rice, E. D. Escarcega, and M. S. Riggs. Renovating secondary sewage by ground water recharge with infiltration basins. Project 16060 DRV, Environmental Protection Agency, March 1972.
5. Gilde, L. C. Poultry processing waste-water treatment system enhances the environment. Amer. Soc. Mech. Engin., Plant Engineering and Maintenance Conference, Fort Worth, Texas, March 1970.
6. Hinesly, T. D., and B. Sosewitz. Digested sludge disposal on crop land. Jour. Water Pollut. Control Fed., 41, 822-830, 1969.
7. Law, J. P., R. E. Thomas, and L. H. Myers. Cannery wastewater treatment by high-rate spray on grassland. Jour. Water Pollut. Control Fed., 42, 1621-1631, 1970.

8. McGauhey, P. H., and R. B. Krone. Soil mantle as a wastewater treatment system. SERL Report 67-11, Univ. of California, Berkeley, 1967.
9. Pennsylvania State University. Proc. of Symposium on Recycling Treated Municipal Sewage and Sludge Through Forest and Cropland. August 1972.
10. Phillipp, A. H. Disposal of insulation board mill effluent by land irrigation. Jour. Water Pollut. Control Fed., 43, 1749-1754, 1971.
11. Porges, R., and G. J. Hopkins. Broad field disposal of beet sugar wastes. Sewage and Indust. Wastes, 27, 1160-1170, 1955.
12. Rice, R. C. Soil clogging during infiltration with secondary sewage effluent. Jour. Water Pollut. Control Fed. (In press)
13. Romero, J. C. The movement of bacteria and viruses through porous media. Ground Water 8, 37-49, 1970.
14. Truesdale, G. A., A. E. Birkbeck, and D. Shaw. A critical examination of some methods of further treatment of effluents from percolating filters. Proc. Inst. of Sewage Purification, Part I, 3-23, 1964.

CROP SELECTION AND MANAGEMENT

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INTRODUCTION

Crops have been grown for centuries on land used for spreading manure and sewage. These materials were regarded as fertilizers, not wastes, and no one considered that their use posed any problems of crop selection and management. Animal manure was spread on food and feed crops, at rates of a few tons per acre. Sewage and human wastes were less often used, particularly in Western countries, but many cities spread sewage, with or without primary treatment, on land used for crops.

Berlin's sewage farm was started about 1850, and by 1905, it covered 21,000 acres and supplied one-fourth of the vegetables for the city's population. Paris established its first farm in 1870. Both these cities, and many others in Europe, grow vegetables, tree fruits, cereals, and forage crops. Melbourne, Australia's Werribee Farm began operation in 1893, and since then has produced 266,000 cattle and nearly 1.5 million sheep on pastures irrigated and fertilized with sewage.

Pasadena, California started a sewage farm in 1887, and by 1935, 90 of the 310 municipalities in California were spreading sewage on land. Vegetables, fruit trees, grapes, alfalfa, sugarbeets, hops,

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and cotton were grown; only salad vegetables and berries were forbidden. San Antonio, Texas started in 1915, growing corn, grain sorghum, cotton, forages, pecans, citrus, vegetables, and roses for cut flowers.

BENEFITS OF CROPPING

Wastes were originally applied to crop land because the nutrients they contained raised crop yields. This is still true; for example, a ton of manure contains \$1-2 worth of plant nutrients. Even in cases where land disposal is not profitable, it reduces the cost of waste disposal. Livestock sold from Melbourne's Werribee Farm paid 12% of the operating costs of the farm in 1971-72. Finally, initial costs of cropland disposal systems may be less than those of other systems. A large canning plant in New York disposes of its effluent by spray irrigation of cropland, at a cost 45% less than that of an extended aeration system.

However, economics are not the only reason for growing crops on waste disposal fields. Crop cover speeds infiltration of water and liquid wastes; reduces soil and nutrient losses through erosion; and reduces organic matter and plant nutrient content of run off and ground water.

The short-term effect of different crops on the rate at which water enters the soil is shown in Figure 1. The data was obtained on three Southeastern Piedmont soils; cropping system effects may be greater on deeper, more porous soils, or considerably less on shallow or less permeable soils. Long-term effects may be even greater on well-drained soils. In a California study on a sandy loam soil, flooded bermudagrass took up 11.6 feet of water per day, versus 3.6 feet taken up by bare soil. At the end of 200 days of flooding, the bare soil surface has become sealed by clay and organic particles, and only 0.3 foot of water per day was taken up. Infiltration was reduced under bermudagrass too, but not nearly as much; uptake was still 5.4 feet per day. When sewage effluent or other liquid wastes are applied, rapid infiltration is essential to achieve maximum loading rates.

Rapid infiltration reduces runoff and soil erosion; this is important whether liquid or solid wastes are applied. The effectiveness of different crops in reducing soil loss varies with soils, topography, stage of growth, and amount and distribution of rainfall. Soil losses under truck crops, corn, small grains, and established pasture are generally about 40-50%, 30%, 25%, and 1-2%, respectively of losses from bare soil. Contouring, by reducing runoff velocity, can reduce soil losses under row-crops by as much as 50% on slopes of 2-7%, and leaving crop residue in the field can reduce losses by retarding runoff and by increasing infiltration capacity.

If soil loss is reduced, so is the phosphate content of runoff water, because most of the phosphate is carried on soil particles. In Alabama, erosion carried off 1430 pounds of phosphate per acre over 26 years; this was 82% of the phosphate applied as fertilizer. In Missouri, continuous corn lost 18 pounds of phosphate per acre per year through erosion, while bluegrass pasture lost only 0.1 pound.

Crops aid in the removal of organic matter and plant nutrient elements from applied wastes. Runoff water from crop and forest land sprayed with food processing plant or sewage effluent shows 70-90% reduction in organic matter and 60-80% reduction in nitrogen. Growing crops removed nutrients from water percolating through the soil or from solid wastes plowed under. This helps prevent contamination of ground water. Table 1 shows estimates of the amounts of nutrients removed by many crops at above average yields. These estimates are based on average composition; heavy applications of wastes may increase the content of certain elements, the crop yield, and the amount removed by the crop. Table 2 shows the increase in the amount of nitrogen removed by seven grasses as nitrogen fertilization is increased.

Harvested forages remove more nutrients than other crops, because all top growth is harvested. However, when forages are used as pasture, most of the nutrients pass through the grazing animal and return to the soil, so net removal is low.

LIMITATIONS TO CROPPING

The rate of nutrient removal by different crops may determine which crops can be grown, and the rate of water application. Crop selection will also be influenced by crop tolerances to waterlogging, salts, and heavy metals. Finally, animal and human health problems must be considered before crops grown on waste disposal fields can be used for feed or food.

On a short term basis, the rate of nitrogen removal by crops may be the limiting factor in waste application rates. A proposed regulation in Ontario would require one acre to dispose of wastes from two dairy cows, while California workers say one acre is required for three cows. These requirements are equivalent to a maximum rate of 30 to 45 tons of manure, or about 300 to 450 pounds of nitrogen, per acre. Iowa and Pennsylvania research indicates 500-600 pounds of nitrogen per acre is the maximum. A Maine study looked at the effects of soil type, depth, and drainage on nitrogen loading rates. These workers recommended no more than 200 pounds of nitrogen per acre on poorly drained soils; 350 pounds on well drained or excessively drained soils; and 700 pounds on soils with a fragipan or claypan. Single applications at these rates, followed by several years without manure, should not raise nitrate content of ground water above the recommended maximum of 10 ppm of nitrate nitrogen. Annual application rates should be no more than half this on non-cropland, but if a crop is grown, these half rates can be increased by the amount of

nitrogen removed by the crop. Application rates for any nitrogen-containing waste can be calculated by dividing the recommended rate of nitrogen application by the nitrogen content of the waste.

It may not be practical to apply these maximum rates of nitrogen. They may cause lodging of small grain crops, reduce the quality of some vegetables, and cause legumes to die out of a grass-legume pasture. However, they should cause no problems on crops which customarily receive high rates of nitrogen, such as corn and the forage grasses.

Grass pastures fertilized with high rates of manure or other wastes may contain high levels of potassium and nitrogen, and low levels of magnesium. Cattle often develop grass tetany on such forages, particularly when cool, cloudy weather follows a warm, sunny period. Tetany usually can be prevented by feeding up to 2 ounces per head per day of a magnesium supplement. It may be necessary to alter time and rate of application when weather conditions associated with tetany occur frequently; research is in progress.

Fat necrosis is a serious problem on tall fescue fertilized with very high rates of manure; half or more of the cattle on such pastures may develop fatty tumors on the intestines. However, this condition seems unique to fescue pastures; other grasses which respond to high nitrogen rates, such as Coastal bermudagrass and reed canary-grass, have not caused this condition. Nitrate poisoning may occur on annual pastures receiving high rates of nitrogen, especially when growth is slowed by cold or drought; avoid grazing during these periods.

Crops which tolerate water-logging of the soil or shallow water tables must be grown on areas where liquid wastes are applied at high rates. Table 3 shows relative yields of many crops at different water table depths. Other studies have shown that reed canarygrass is probably the most water-tolerant forage, and that birdsfoot trefoil, alsike clover, and Coastal bermudagrass are highly water-tolerant. Rates of liquid waste application are more likely to be determined by site considerations (Chapters 1 and 9) than by crop tolerances, but once the rate is fixed it may be necessary to consider these tolerances.

Many waste materials may contain large amounts of soluble salts. Continued application of these wastes over a period of time could result in an accumulation of salts in the soil rooting zone. The degree to which this occurs will depend upon the amount of salt applied in the waste and the degree of leaching that occurs as a result of applied irrigation and rainfall. The actual effect of salinity on plant growth is determined by the salt concentration of the soil solution in the root zone. For a given amount of salt in the soil, the concentration in the soil solution would depend on the soil water content, being high in a relatively dry soil and low in a wet one.

A more detailed discussion of salinity, including data on salt tolerance of crops, is presented in a previous section. Several factors regarding plant effects should be stressed, however. Salt tolerance may vary with stage of growth, the seedling stage usually being the most sensitive. In some crops, the vegetative portion of the plant may be more affected by salinity than the fruit, and vice versa. Since a major effect of salinity results from a physiological drought, greater damage occurs when

evapotranspiration is high. Under such conditions, sprinkler-irrigating with a saline waste may produce a foliar buildup of salts great enough to cause burning.

Salt tolerances of different crops can be estimated from Figure 1. Varietal differences also exist. For example, Coastal and Suwannee bermudagrasses can tolerate up to 40% more salt than common, Greenfield and NK-37 bermudas. Specific ion effects are usually negligible but must be considered. For example, some crops such as stone fruits, citrus, avocados, and berries are especially sensitive to high chloride concentrations. Under certain conditions, it is possible that the composition of a saline waste may create a nutritional imbalance in the soil. Although few examples of this have been identified, it is a factor to be considered.

Food processing wastes and some industrial effluents may be very acid or alkaline. Soils with a high clay or organic matter content have a high buffering capacity, and can absorb sizeable amounts of such materials without a significant change in their acidity or alkalinity. However, it is often best to neutralize excessively acid or alkaline wastes before application.

Boron is an essential plant nutrient, but it can cause crop damage, even at the very low concentrations encountered in some wastes. Since borax is a common cleaning agent, it is possible that certain municipal, processing plant and other effluents may have a sufficiently high content of boron to affect plant growth. Crops which can tolerate different levels of boron in irrigation water are shown in Table 4; those at the top of each column are more tolerant than those at the bottom.

Sewage sludges, some manures, and industrial wastes may contain large amounts of heavy metals, particularly copper, zinc and nickel (Chapters 2 and 6). These metals will probably limit the amount of sewage sludge that can be applied on a long-term basis, because once they are added to the soil they are removed very slowly. On land in England and Germany, which has been fertilized with sewage sludge and effluent for several decades, crop yields have begun to decline because of copper and zinc accumulation. This decline may be caused by a nutrient imbalance, rather than actual toxicity. For example, high zinc levels in the soil inhibit manganese uptake, but this can be corrected by applying manganese fertilizer. However, it has been suggested that the total amount of copper, nickel and zinc added to a soil over a time period should not exceed 500 pounds per acre of zinc plus "zinc equivalent." "Zinc equivalent" is calculated by multiplying copper content by two and nickel content by eight, because these two metals are approximately two and eight times, respectively, as toxic as zinc.

Crop species and varieties differ in tolerance to these metals (Table 5), but tolerances are not as well defined as the tolerances to salts and boron. Metal toxicity is less severe on soils with a high content of clay or organic matter, and on neutral or slightly alkaline soils. This suggests that application of lime or manure would help prevent toxicity, but sometimes these materials themselves have a high metal content. Considerable amounts of zinc may occur in limestone, and manure may be high in copper derived from feed additives.

Crops fertilized with metal-containing wastes may accumulate these metals in the portions harvested for food and feed. These accumulations might be toxic, or might exceed present tolerances prescribed by EPA for agricultural commodities (Table 6). These tolerances apply only to heavy metals derived from pesticides, and represent the minimum practical level consistent with the use of a particular pesticide, at a particular stage of growth. For example, pears sprayed with copper carbonate after harvest may not contain more than 3 ppm copper, but there is no limit on copper content of pears sprayed before harvest. Unfortunately, chemical analysis cannot distinguish whether copper was applied before or after harvest, or came from sewage sludge applied to the soil. Uniform tolerances must be devised for those elements likely to accumulate in crops fertilized with wastes. FDA is determining permissible levels of mercury, copper, zinc, cadmium, arsenic, and selenium in food, regardless of source.

RECOMMENDATIONS

Effluents and other waste waters may be applied to crops by standard irrigation techniques. Amounts may be in excess of usual irrigation rates. The limiting factors will be the capacity of the soil to retain waste components that must be kept out of the ground water, and the infiltration rate of the soil. Infiltration rates are given in local Conservation Irrigation Guides, available from the Soil Conservation Service of the USDA. These guides also contain information on irrigation frequency and water needs of different crops.

Solid wastes may be spread on the surface or incorporated. However, surface spreading greatly increases the risk of contaminating surface water through runoff. To minimize this risk, do not spread on snow or frozen ground if the slope is greater than 3%, and do not spread at any time on slopes greater than 25%. Do not spread within 100 feet of any well or standing body of water, or within 25 feet of the high-water mark of any watercourse. Grass buffer strips along the edge of bodies of water will reduce the risk of contamination. Solid wastes can be spread on the surface and plowed under; spread in furrows and covered; or applied with special equipment which lifts a slice of soil, injects the material, and replaces the soil. Crop tolerances to salt, nitrate, and heavy metals may limit the amount which can be disposed of by surface application or shallow incorporation. With deep burial of wastes, applications must be limited to rates which will contaminate ground water.

Freshly applied manure or sludge may inhibit germination. It is best to wait a week or two before planting, to allow some leaching of salts and nitrates, and release of ammonia and other volatile materials, before seeding.

Crops must be chosen which are adopted to local climatic conditions and high rates of fertilization, as well as to special conditions imposed by the characteristics of the waste applied. It is absolutely necessary that you know the range of pH, nitrogen, salt, and heavy metal content, as well as the average value, to select adapted species and varieties. Harvested crops remove more of the applied nutrient elements from the soil than grazed crops, but require more machine operations, which may interfere with disposal. Perennial crops may be grown where liquid wastes are applied or solid wastes are spread on the surface, but solid wastes cannot be incorporated without damaging the stand. Perennial crop stands may require periodic renovation to restore high infiltration rates. In one case, infiltration rate on reed canarygrass decreased from 10 feet per day to 0.4 foot in six years, and the decrease on bermudagrass has been mentioned.

Because of the great diversity in waste material composition, crop tolerances, and soil and climatic conditions, it is not possible to make specific crop and management recommendations. We are also hampered by a lack of information on long-term effects of waste disposal on crop production. However, the guidelines we have proposed

should help you select the combination of crop and management practices best suited to your situation. Two examples will show how the guidelines might be applied, and how results actually obtained compare with those predicted.

The Campbell Soup Co. plant in Paris, Texas used wastewater to irrigate reed canarygrass grown for hay. The water contained 17.4 ppm nitrogen, or 47.2 pound per acre-foot (1 acre-foot of water = 2.72 million pounds). The soil is a poorly-drained clay loam; on such soils, the recommended N rate is 100 pounds, plus the 169 pounds removed by the crop (Table 1), or 269 pounds per acre per year. Dividing 269 by 47.3 we get a maximum rate of 5.7 feet of water per year. The plant operates 43 weeks a year; if we irrigate 8 hours a day, 5 days a week, we can apply .32 inches per day or .04 inches per hour, just under the infiltration capacity of the soil. At this rate, runoff and nitrate content of the ~~runoff~~ runoff should be minimal. The company actually applied 10 feet of water, containing 473 pounds of N. Runoff amounted to 61% of the total water applied (irrigation plus rain), and the runoff contained 45 pounds of N. Ten feet of water was too much for the soil to handle. However, reed canarygrass was the right crop to grow, because its tolerance to water-logging enabled it to grow well under these conditions.

In a Pennsylvania study, manure containing 12.55 dry matter and 2.35% N in the dry matter was applied to orchardgrass growing on a well-drained silt loam. Well-fertilized orchardgrass should yield

about 4 tons per acre in Pennsylvania, and contain 18% protein or 2.9% N, for a total N removal of 232 pounds per acre. To this, add the Maine recommendation of 175 pounds per acre for well-drained soils, for a total application of 407 pounds of nitrogen, 8.7 tons manure dry matter, or 70 tons per acre of manure as it comes from the barn. However, the lowest rate actually applied was 15 tons of dry matter, containing 705 pounds of N, and the orchardgrass yielded only 2.6 tons per acre. As a result, nitrate content of the soil water went up to 13 ppm, above the recommended 10 ppm maximum. Manure rates as high as those in the study should not be used on orchardgrass.

Of necessity, this handbook can make only general recommendations for the application of waste materials to farm land. Furthermore, recommendations may change as more information becomes available. Therefore, for the most complete, up-to-date information for your particular location and cropping program, consult the specialists of your State and Federal Extension Service and the Soil Conservation Service.

REFERENCES

- American Society of Agricultural Engineers. 1971. Livestock waste management and pollution abatement. Proceedings of the International Symposium on Livestock Wastes. ASAE Publ. PROC-271. 360 p.
- Chumbley, C. G. 1971. Permissible levels of toxic metals in sewage used on agricultural land. ADAS Adv. Paper No. 10. 12 p.
- Cornell University. 1969, 1970, 1971, 1972. Conferences on Agricultural Waste Management. Cornell University, Ithaca, New York.
- Council of State Governments. 1972. Animal Waste Management. Proceedings of the National Symposium on Animal Waste Management. Council of State Governments, Washington, D. C.
- Lagerwerff, J. V. 1967. Heavy-metal contamination of soils. p. 343-364. In N. C. Brady, ed., Agriculture and the quality of our environment. AAAS Publ. 85.
- Lyerly, P. J., and D. E. Longnecker. 1957. Salinity control in irrigation agriculture. Texas Agr. Exp. Sta. Bull. 876, 20 p.
- Reed, S. (coord.) 1972. Wastewater management by disposal on the land. Cold Reg. Res. Eng. Lab. Spec. Rep. 171. 184 p.
- Richards, L. A. (ed.) 1954. Diagnosis and improvement of saline and alkali soils. USDA Agr. Handbook 60. 160 p.
- University of Maine. 1972. Maine guidelines for manure and manure sludge disposal on land. Maine Agr. Exp. Sta. Misc. Rep. 142. 21 p.
- Wadleigh, C. H. 1968. Wastes in relation to agriculture and forestry. USDA Misc. Publ. 1065. 112 p.

Table 1. Removal of different elements from soils by crops and animals.

Crop	Yield per acre	Pounds removed per acre						Grams removed per acre				
		N	P	K	Ca	Mg	S	Na	Fe	Mn	Cu	Zn
Corn grain	100 bu	80	15	17	2	8	7	1	66	14	6	43
Grain sorghum	80 bu	80	14	15	2	8	7	2	90	30	20	28
Soybeans	32 bu	105	11	29	5	5	4	4	70	26	14	--
Peanuts	2500 lb	94	8	12	2	4	6	14	16	--	--	--
Cottonseed	1800 lb	62	13	20	3	6	4	5	114	10	41	--
Wheat	60 bu	81	15	18	2	4	6	3	73	80	11	23
Rice	6000 lb	78	14	9	3	4	3	3	96	48	9	5
Barley	75 bu	67	15	20	3	5	6	1	87	26	12	25
Sugarbeets	25 T	21	20	125	20	15	1	40	227	765	30	--
Corn silage	20 T	136	24	118	34	24	12	3	929	228	47	98
Alfalfa hay	7 T	332	31	212	197	38	43	19	1306	282	74	92
Coastal bermuda hay	9.5 T	243	29	270	74	27	--	--	--	--	--	--
Reed canarygrass hay	7 T	169	30	282	41	31	--	47	816	503	65	--
Potatoes	30 T	210	30	288	6	18	12	12	544	240	102	--
Tomatoes	20 T	71	11	98	5	6	--	1	92	--	--	--
Lettuce	12.5 T	34	5	42	5	3	--	2	55	--	--	--
Carrots	20 T	58	12	112	12	8	8	15	104	68	24	--
Snap beans	5 T	27	4	21	5	3	--	1	32	--	--	--
Dry beans	1800 lb	64	8	22	3	3	4	1	64	15	8	--
Loblolly pine	Annual growth	9	1	4	5	2	1	--	--	--	--	--
Beef gain	500 lb	10	4	1	8	1	1	1	46	--	--	--
Milk	4000 lb	22	4	6	5	1	1	2	1	1	--	--

References:

- Miller, D. F. 1958. Composition of cereal grains and forages. NAS-NRC Publ. 585. 663 p.
 Morrison, F. B. 1951. Feeds and feeding, 21st ed. Morrison Publ. Co., Ithaca, N.Y. 1207 p.
 Switzer, G. L., and L. E. Nelson. 1972. Nutrient accumulation and cycling in loblolly pine (*Pinus taeda* L.) plantation ecosystems: The first twenty years. Soil Sci. Soc. Amer. Proc. 36:143-147.
 Watt, B. K., and A. L. Merill. 1963. Composition of foods. USDA Agr. Handb. 8. 190 p.

Table 2. Nitrogen removal by grasses at different rates of nitrogen fertilization.

Grass	Pounds N per acre applied				
	80	160	320	480	640
	Pounds N per acre removed				
Orchardgrass	31	75	115	139	147
Bromegrass	47	104	166	204	217
Tall oatgrass	45	80	96	115	122
Intermediate wheatgrass	46	101	176	213	227
Tall wheatgrass	57	94	141	163	160
Tall fescue	50	67	114	133	127
Coastal bermudagrass	68	115	210	292	348

References:

- Burton, G. W., J. E. Jackson and R. H. Hart. 1963. Effects of cutting frequency and nitrogen on yield; in vitro digestibility, and protein, fiber, and carotene content of Coastal bermudagrass. Agron. J. 55: 500-502.
- Dotzenko, A, D. 1961. Effect of different nitrogen levels on the yield, total nitrogen content, and nitrogen recovery of six grasses grown under irrigation. Agron. J. 53:131-133.

Table 3. Relative yields of crops at varying water table depths.

Crop	Approximate water table depth, cm										Soil type
	15	30	45	60	75	90	102	120	150	240	
Corn	30 80 45	100	85 96 67	83 70	100	93 100					Fine sandy loam Loamy fine sand Loam & silty clay loam
Millet	41 57	69	80 77	87	98 100	100	93				Loam Fine sandy loam
Soybeans	66		100		82						Fine sandy loam
Wheat			58	77		89		95	100		Clay
Barley			58	80		89		95	100		Clay
Oats			49	74		85		95	100		Clay
Sugarbeets			71	84		92		97	100		Clay
Orchardgrass	28	100	93								Silt loam
Tall fescue	51	100	87								Silt loam
Ladino clover	100	97	88								Silt loam
Alfalfa				100				97		96	Clay loam
Potatoes			90	100		95		92	96		Clay
Beans	71 100 28 1 74	78 83 43 52	79 80 89 100 73 79	84 94 94 78 97		90 100 92 75		94	100		Clay Silty clay loam Loam Loamy fine sand Fine sandy loam Fine sandy loam

Table 3. (cont.)

Crop	Approximate water table depth, cm										Soil type
	15	30	45	60	75	90	102	120	150	240	
Cabbage	33	100	97	100	98						Fine sandy loam
	6	85	84	89	89	100	97				Loam
	54		91		100						Fine sandy loam
Squash	2	49	65	100	97						Fine sandy loam
	21	48	58	65	78	90	100				Loam
Mustard	52	96	100	93	95	97	99				Loam
Pepper	80		86		100						Fine sandy loam
Peas			50	90		100		100	100		Clay
Tomato	9	28	47	60		100					Silty clay loam, loam, & loamy fine sand

References:

- Williamson, R. E., and J. R. Carreker. 1970. Effect of water-table levels on evapotranspiration and crop yield. Trans. ASAE 13:168-170, 176.
- Williamson, R. E., and G. J. Kriz. 1970. Response of agricultural crops to flooding, depth-of-water table and soil gaseous composition. Trans. ASAE 13:216-220.

Table 4. Crop tolerances to boron in irrigation water; crops at top of each column are more tolerant than those at bottom (From Eaton, F. M. 1935. Boron in soils and irrigation waters and its effect on plants with particular reference to the San Joaquin Valley of California. USDA Tech. Bull. 448. 131 p.)

Tolerant (1-3 ppm B)	Semi-tolerant (.67-2 ppm B)	Sensitive (Less than 1 ppm B)
Asparagus	Sunflower	Pecan
Date palm	Potato	Black & English walnut
Sugarbeet	Cotton	Jerusalem artichoke
Stock & garden beet	Radish	Navy bean
Alfalfa	Field pea	Plum
Broad bean	Olive	Pear
Onion	Barley	Apple
Turnip	Wheat	Grape (Sultanina, Malaga)
Cabbage	Corn	Kadota fig
Lettuce	Sorghum	Persimmon
Carrot	Oats	Cherry
	Pumpkin	Peach
	Tomato	Apricot
	Bell pepper	Blackberry
	Sweet potato	Orange
	Lima bean	Avocado
		Grapefruit
		Lemon

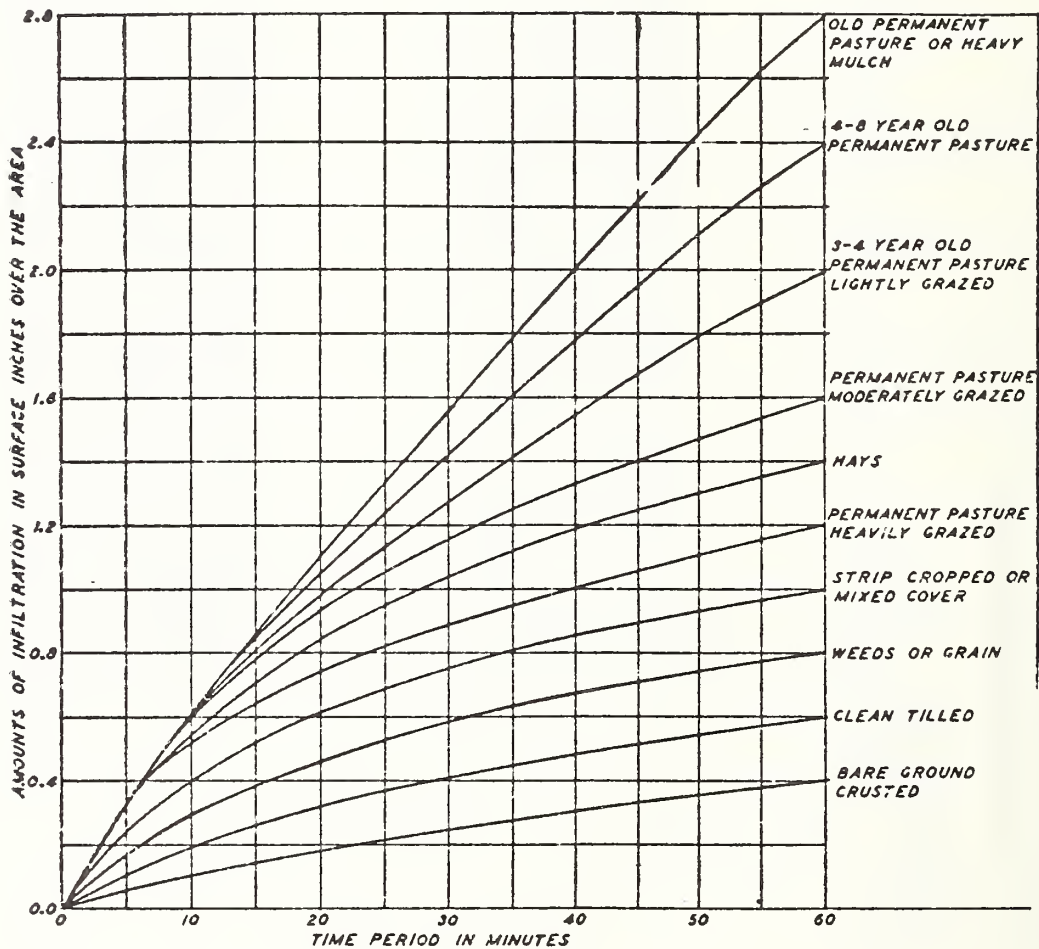
Table 5. Crop tolerance to copper, zinc, and nickel, in order of decreasing tolerance.

Copper	Zinc	Nickel
Bermudagrass,	Bermudagrass, rye-	Bermudagrass
orchardgrass,	grass, orchard-	Barley, wheat
ryegrass	grass, bentgrass	Ryegrass
	Corn, oats, wheat,	
	rye	Beans
Oats, beans	Birdsfoot trefoil	Oats, corn
	Potatoes	Tobacco
	Beans, cowpeas	
Citrus, celery	Turnips, carrots	Red clover,
	Buckwheat	potato, orange
	Cabbage	
Beets, turnips	Lettuce	Turnips,
	Beets	cabbage, kale
Spinach	Spinach	Beets

Table 6. Heavy metal and boron tolerances in or on raw agricultural commodities. (From Code of federal regulations, 1972. Title 40, Protection of environment. Chapter 1, Environmental Protection Agency. Part 180, Tolerances and exemption from tolerances for pesticide chemicals in or on raw agricultural commodities. Genl. Svcs. Adm.)

Element	Tolerance, ppm (fresh weight)	Product	Source
Copper	3.0	Pears	Basic copper carbonate, post harvest
Zinc	30.0	Peaches	Basic zinc sulfate
Lead	7.0	Apples, apricots, asparagus, avocados, blackberries, boysenberries, celery, cherries, currants, dewberries, eggplants, gooseberries, grapes, loganberries, mangoes, nectarines, peaches, pears, plums, quinces, raspberries, strawberries, tomatoes, youngberries	Lead arsenate
Arsenic (as As_2O_3)	1.0	Citrus fruits	Lead arsenate
	0.7	Cottonseed	Methanearsonic acid
	4.0	Cottonseed	Orthoarsenic acid
Boron	3.5	Asparagus, beans, blackberries, blueberries, boysenberries, broccoli, brussels sprouts, cabbage, carrots, cauliflower, celery, collards, corn, cucumbers, dewberries, eggplants, grapes, kale, kohlrabi, loganberries, melons, peppers, pumpkins, raspberries, rutabagas, spinach, squash, strawberries, summer squash, tomatoes, turnips, turnip greens, youngberries	Calcium copper, magnesium, or sodium arsenate
	30.0	Cottonseed	Sodium borate
	8.0	Citrus fruits	Borax or boric acid

Figure 1. Infiltration rates under different crops (From Holtan, H. N., and M. H. Kirkpatrick, Jr. 1950. Rainfall, infiltration, and hydraulics of flow in run-off computation. Trans. Amer. Geophys. Union 31:771-779).



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